Applying an ecosystem service approach to floodplain habitat restoration

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Abstract

The concept of ecosystem services can provide a framework for holistic habitat management, but has rarely been applied at local scales. In recent history floodplain management in the UK has focused primarily on agriculture and flood defence. Increasingly, floodplains are being managed for a broader range of ecosystem services, for example by reinstating flooding regimes. This study investigated the relationships between floodplain management and service provision in a newly restored floodplain in South Yorkshire, and assessed the utility of an ecosystem service framework for habitat management.

Two ecological impacts of hydrological restoration were analysed: plant community change, and water vole persistence. There were hydrologically driven changes in plant community composition, but colonisation by new species was low and there was no evidence of the formation of floodplain grazing marsh target communities. Water voles were more likely to occur around wider water bodies, in areas of tall, diverse vegetation. The restoration of flooding did not negatively impact the water vole distribution.

A framework for using encounters with different habitat elements to model recreational experiences at local scales was developed, and was used to compare the quality of recreational experiences provided in different parts of the floodplain.

Examination of the relationships between floodplain heterogeneity and the provision of multiple ecosystem services revealed that more heterogeneous floodplains tended to be suboptimal in terms of delivering particular ecosystem services, but more evenly balanced the provision of different services.

The potential for an ecosystem service framework to affect habitat management decision making was tested using a decision making exercise. Participants who were provided with information on the greatest number of ecosystem services showed more variable preferences for management scenarios, which could encourage multifunctional management.

The findings of the study as a whole suggest that ecosystem service frameworks could feasibly be applied to local habitat management.

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Declaration

I wrote all chapters of this thesis, but the work has been developed following discussions with Phil Warren, Lorraine Maltby, and Helen Moggridge. Together we identified the research questions, planned the field sampling, and discussed data analysis. I collected all field and survey data except the baseline plant community dataset used in Chapter 3, the baseline water vole distribution records used in Chapter 4, and the vegetation height dataset used in Chapter 5. I conducted all relevant laboratory work and data analysis, and drafted and made alterations to the thesis. Chapters 4, 5, 6, and 7 have been prepared as manuscripts for submission to particular journals, and DR, PW, LM, and HM will be the named authors on all of the resulting publications. DR will be the first author in all cases.

Related publications

Chapter 4 of this thesis has been accepted for publication at Wetlands Ecology and Management. The current citation is;

Richards D, Maltby LM, Moggridge HL, Warren PW (in press) European water voles in a reconnected lowland river floodplain: habitat preferences and distribution patterns following the restoration of flooding. Wetl Ecol Manag.

This project inspired an additional manuscript in a related area, which has now been published. This publication is not part of the thesis:

Richards DR (2013) The content of historical books as an indicator of past interest in environmental issues. Biodivers Conserv 22:2795-2803.

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1. Introduction

Humans actively manage between 39 and 50% of the world's terrestrial habitats (Vitousek et al. 2010). Habitats can provide many goods and services that allow and enhance human survival, but there are commonly trade-offs between the provision of these different benefits (Rodríguez et al. 2006; Bennett et al. 2009). Habitat management is therefore a balancing act, and compromises must be made between different human demands (Rouquette et al. 2011). The pressure of human demands on habitats is likely to increase in the future (Malmqvist and Rundle 2002), so it is critical that habitat management is designed with consideration of a broad range of these demands (Tallis and Polasky 2009). The concept of ecosystem services can be used as a framework to consider the range of benefits that habitats can provide, but practical implementation of ecosystem service frameworks for habitat management has been slow (Turner and Daily 2008). This study aims to inform the implementation of ecosystem service frameworks, by focusing on the management of lowland river floodplains. This introductory chapter first defines an ecosystem service framework for habitat management, and then reviews some of the gaps in knowledge which currently constrain implementation. It then outlines the pressures facing floodplain systems and the research opportunity that they provide, and summarises the objectives of the following chapters and the study as a whole.

1.1. An ecosystem service framework for habitat management

The term "habitat" has two broad meanings (Miller and Hobbs 2007). A habitat can be defined as the area that supports a particular species (Hall et al. 1997), or can refer to "habitat types" (Daubenmire 1968); areas that have similar environmental conditions and support similar communities of organisms (Miller 2000; Miller and Hobbs 2007). In this study the latter definition is used when discussing "habitat management" because it is the response of multiple factors that are of interest, rather than the conservation of one species. Habitat management is the manipulation of habitats by humans, and the definition used here is similar to the definition of "ecosystem management" (Grumbine 1994), in that it includes any manipulation of the physical, ecological, or social components of the system. Examples of management actions include vegetation cutting (Buttler 1992), altering cattle stocking densities (Jansen and Roberston 2001) or manipulating ground water levels (van Bodegom et al. 2006). Habitat management in the United Kingdom (UK) is commonly planned locally because individual landowners or organisations own relatively small areas; the mean area of a farm is 70 ha (Eastwood et al. 2010), the median area of protected area is 110 ha (Oldfield et al. 2004), and the median area of a Site of Special Scientific Interest (sites designated by a UK government agency for their wildlife or geology) is 20 hectares (Oldfield et al. 2004).

Ecosystem services are the benefits that people gain from habitats (MA 2005; Fisher et al. 2009). This study considers three categories of ecosystem service as defined by the Millennium Ecosystem Assessment; provisioning, cultural, and regulating services (MA 2005). Provisioning ecosystem services are generally physical goods that are required for human survival, such as food or fuel (Fisher et al. 2009). Cultural ecosystem services are those that contribute to psychological or social wellbeing, such as recreation or cultural identity (Chan et al. 2012; Daniel et al. 2012). Regulatory ecosystem services are mechanisms or processes that regulate the environment and make it suitable for habitation, for example erosion control or carbon dioxide regulation (Orr et al. 2008). Supporting services, which were the fourth type of ecosystem service classified by the Millennium Ecosystem Assessment, are not considered as services in this study. Supporting services are the processes that are necessary for the provision of other services, such as nutrient cycling and pollination (MA 2005). Categorising supporting services can lead to confusion, because their importance is recognised twice, both at the “supporting” stage, and as the provisioning, regulatory or cultural service that is provided to people (Hein et al. 2006; Bo-Jie et al. 2011). The ecological and physical processes that underlie ecosystem service provision are instead referred to as “functions” or “processes” in this study (Daily 1997). The parts of ecosystems that contribute directly to provide services, such as the organisms that provide food or aesthetic value (de Groot et al. 2002; Fisher et al. 2009; Haines-Young and Potschin 2010) are referred to as “components” or “indicators” of ecosystem services (see Figure 1.1 for a conceptual diagram of the links between habitat conditions and ecosystem service provision).

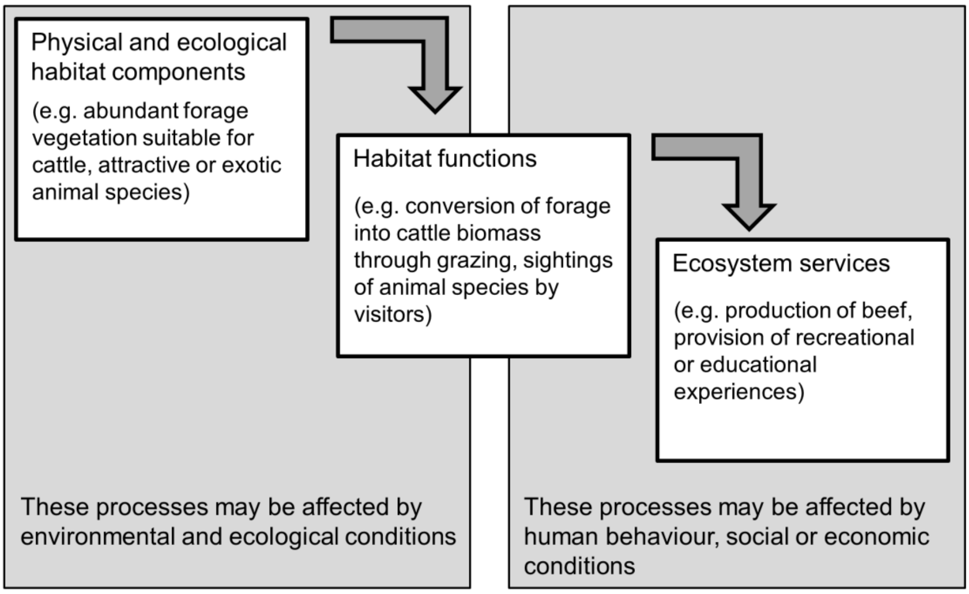

Figure 1.1. The physical and ecological components of habitats can perform or contribute to habitat functions. Ecosystem services are provided when people make use of the components and functions of habitats. Based on Figure 2 in Haines-Young and Potschin (2010).

A habitat can provide a number of ecosystem services, but there are commonly trade-offs between their provision (Rodríguez et al. 2006; Bennett et al. 2009; Raudsepp-Hearne et al. 2010a). The suite of ecosystem services that a habitat provides depends on the way that it is managed, so habitat management can be designed to target a suite of services that is most desirable (Tallis and Polasky 2009; de Groot et al. 2010). In the past, habitats have been managed for a narrow range of services (Tallis and Polasky 2009), commonly for provisioning services (Raudsepp-Hearne et al. 2010b). These ecosystem services have been prioritised above others because the demand for them is clearly apparent, for example, food and fuel are needed for survival on a daily basis and have economic values (MA 2005). In contrast, the value of the other ecosystem services has historically been less obvious (Tallis and Polasky 2009; Laurans and Mermet in press). For example, the importance of atmospheric carbon regulation has only been widely realised over the past 30 years (Demeritt 2001). Past habitat management has commonly led to the degradation of less noticeable ecosystem services, because they have not been considered during the decision making process (Tallis et al. 2008). The concept of ecosystem services provides a framework for recognising a wider range of benefits that habitats can provide (Fish 2011). An ecosystem service framework should allow people to make more informed decisions about different habitat management options, by describing the trade-offs and synergies between the provision of different services (Tallis and Polasky 2009; Haines-Young and Potschin 2010).

Ecosystem service frameworks have been controversial (Boyd and Banzhaf 2007; Fisher et al. 2009), so it is necessary to clarify some of the details of the definition used in the present study. First, there has been much debate about the monetisation of ecosystem services (Armsworth et al. 2007). Ecosystem services were originally monetised to convey the importance of natural ecosystems to non-ecologists (Gómez-baggethun et al. 2010). More recently, monetary values have been attached to ecosystem services with the hope that this will allow them to be captured by economic markets, for example through payments for ecosystem service schemes (Wunder 2007; Jack et al. 2008). The purpose of this study is to inform habitat management decision making, not to attract a wider audience for conservation or provide a mechanism for the benefits ecosystem services provide to be captured by individuals. It is therefore not useful to monetise the ecosystem services used in this study. Additionally, nonmonetary quantifications of ecosystem services have some advantages over monetary values. Nonmonetary ecosystem service values can be easier for the public to interpret than monetary valuations, particularly for cultural services that are far removed from economic markets (Chan et al. 2012). For example, a wildlife enthusiast may relate better to a list of species that they would be likely to see on a day trip than the equivalent derived monetary value. Monetisation of ecosystem services can also add a layer of complexity to an ecosystem service analysis which can increase the uncertainty surrounding the estimate of their value (Spangenberg and Settele 2010).

There has been some disagreement over whether biodiversity is an ecosystem service in itself, or is an underlying provider of ecosystem services (Turner and Daily 2008; Mace et al. 2012). In this study biodiversity is defined as a cultural ecosystem service, because society values the existence of plants and wildlife (Aldred 1994; Oksanen 1997). In the UK the cultural service provided by biodiversity is reinforced by legislation and government guidance, for example by protected areas, protected species, or biodiversity action plan priority habitat status (Hernández-Morcillo et al. 2013, JNCC 1995).

The negative impacts that habitats can have on human well-being, such as maintaining populations of disease vectors or pest species, have commonly been ignored in ecosystem service research (Dunn 2010). This can lead to bias when comparing the net utility provided by different habitat types (Dunn 2010). This study attempts to avoid bias by considering both the negative and positive effects of habitats on people (see Chapter 5 for an example in relation to recreational ecosystem services), but in the UK many of the most important characteristics of habitats are beneficial. The “disservices” (Dunn 2010) with the greatest impacts on human well-being, such as human predation and the transmission of diseases, are less of an issue now in the UK than they were in the past (Wilson 2004).

An ecosystem service framework for habitat management should be a neutral tool that allows people to make more informed management decisions. However, it is useful to understand why the framework has been developed, and how its proponents hope that it will change habitat management practices. The ecosystem service concept has been interpreted in a number of different ways. The majority of ecosystem service research has been published in the subject areas of ecology and environmental science (Schaich et al. 2010a), and perhaps as a consequence, conservation or restoration of natural habitats is commonly an explicit goal of ecosystem service frameworks (Armsworth et al. 2007; Goldman et al. 2008; Daily et al. 2009). Other interpretations of ecosystem service frameworks are more utilitarian, for example those that aim to optimise the provision of multiple services, and thus reduce inefficiency in net service provision (Lautenbach et al. 2010; Sanon et al. 2012; Seppelt et al. 2013). Studies that attempt to place a monetary value on ecosystem services may fall into this utilitarian category, as monetary values can be used to choose the management option that will give the greatest net economic return (Newton et al. 2012). A further interpretation of ecosystem service frameworks is that they should encourage "multifunctional" habitat management, meaning that habitats should provide a broad range of ecosystem services (Wiggering et al. 2006). Related to the concept of multifunctionality, some interpretations have aimed to alter the social aspects of decision making processes. It has been suggested that decision making under an ecosystem service framework should be collaborative, and should aim to make compromises that satisfy the demands of the people who have an interest in the relevant ecosystem services (Willemen et al. 2010; Rouquette et al. 2011). It is likely that a combination of the goals set by different interpretations of the ecosystem services concept will be desirable for the management of habitats in the future.

1.2. Knowledge gaps in the implementation of ecosystem services as a habitat management framework

The concept of ecosystem services has been described as the "last, best hope" for changing the way that habitats are managed (Daily et al. 2009). Ecosystem services has grown rapidly as a research area (Fisher et al. 2009), and ecosystem service policies have been adopted by international bodies (European Parliament 2012) and governments (Maltby et al. 2011; Fish and Haines-Young 2011). However, there are still considerable gaps in knowledge that limit the application of ecosystem services as a framework for managing habitats (Turner and Daily 2008; Daily and Matson 2008), particularly at small spatial scales (Broekx et al. 2013).

1.2.1. Previous research into ecosystem services as a framework for habitat management

A focus of previous research has been mapping and quantifying the provision of suites of services over large areas (Kremen and Ostfeld 2005). Ecosystem service provision has been analysed across landscapes (Gulickx et al. 2013), regions (Chan et al. 2006; Nelson et al. 2009), provinces (Raudsepp-Hearne et al. 2010a), countries (Maltby et al. 2011; Bateman et al. 2013), and the world (Costanza et al. 1997; Sutton and Costanza 2002). In contrast, most habitat decisions are made at local scales, at sites that are less than 150 ha in area (Oldfield et al. 2004; Eastwood et al. 2010). Studies at large spatial scales have typically analysed ecosystem service provision at coarse habitat resolutions; two thirds of the studies reviewed by (Seppelt et al. 2011) categorised habitat according to land-use categories or broad habitat types. Broad habitat classifications are rarely useful when designing habitat management at a local scale, because opportunities to convert one habitat type to another are rare (Holmes and Nielsen, 1998; Rohde et al. 2006). Local habitat management more commonly involves altering environmental conditions within a broad habitat type (Palmer et al. 1997; Hilderbrand et al. 2005), so requires a more detailed understanding of the links between environmental conditions and ecosystem service provision.

There are some detailed studies of ecosystem services that have analysed provision in relation to environmental or habitat management conditions (Brönmark and Hansson 2002; Grossmann 2012), but these studies have commonly focused on only one or two services rather than a broader suite (Daily et al. 2009; Seppelt et al. 2011). An understanding of the provision of specific ecosystem services can allow habitat management to target their production, but this approach does not fulfil the objectives of an ecosystem service framework; habitat managers are unable to consider the impacts of management on a broad range of ecosystem services, or balance the trade-offs between different services (Nelson et al. 2009). Increasingly, modelling studies have attempted to combine a broad approach with a detailed understanding of each service, to understand the provision of multiple services in detail (Chan et al. 2006; Nelson et al. 2009; Guerry et al. 2012). Modelling frameworks such as InVEST (Nelson et al. 2009; Tallis and Polasky 2009; Guerry et al. 2012), Natuurwaardeverkenner (Broekx et al. 2013), and EcoServ (Durham Wildlife Trust 2014) have combined environmental, ecological, and socioeconomic data to predict the suites of services provided by landscapes. However, while these tools consider the effects of some environmental variables such as topography, they generally compare ecosystem service provision between, rather than within, broad habitat types. For example, a key data requirement for the InVEST suite of models is a land use dataset (InVEST 2014). As discussed above, conversion between broad habitat types is rarely feasible for habitat management at a local scale, so existing tools are more appropriate for considering management at landscape and regional scales. To provide useful information for managing habitats at local scales, the responses of multiple ecosystem services to management and environmental gradients must be understood (de Vos et al. 2010).

The functional assessment approach has been proposed to provide a rapid and easy-to-use tool for evaluating habitat "functions" (i.e. ecosystem services) (Maltby 2009b). The functional approach analyses service provision in relation to environmental units; the application for wetlands requires the hydrogeomorphic units within a site to be identified (Maltby 2009b). The functional approach provides a framework for assessing a number of processes that underlie the provision of certain ecosystem services, but is intended as an early stage in the assessment process rather than an in-depth analysis (Maltby 2009b). Functions are assessed on a qualitative scale that records the likelihood that the function is being performed, rather than as a fully quantitive measurement. More detailed studies that analyse the quantity of provision of each service are required to extend the ecosystem service approach and better inform habitat management decision making.

1.2.2. Specific knowledge gaps addressed by this study

Many ecosystem services are provided by ecological processes or the ecological components of habitats (Haines-Young and Potschin 2010; Bastian et al. 2012), so it is important to understand how habitat management affects individual species and ecological communities (Kremen 2005. The ecological impacts of many habitat management practices are not well documented, for example, habitat restoration projects commonly make substantial changes to environmental conditions (Palmer et al. 1997; Hilderbrand et al. 2005), but only a third of river restoration projects in the United States have been monitored (Bernhardt et al. 2007). The impacts of changes in habitat management can be predicted if the habitat and resource requirements of a species or community are known (Wheeler et al. 2004), so spatial variation across environmental gradients can be used as a proxy for management actions (Toogood and Joyce 2009). However, the creation of suitable habitat conditions is no guarantee that the desired species or community will establish (Hilderbrand et al. 2005), because trajectories of change can be slow (Donath et al. 2003; Bissels et al. 2004) and unpredictable (Trowbridge 2007; Matthews and Spyreas 2010). It is therefore important to analyse the responses of communities and species to changes in habitat management over time (Palmer et al. 2007). An understanding of the mechanisms behind ecological changes may provide insights that can inform the design of future management (Walters and Holling 1990; Bayley 1995; Michener 1997; Palmer et al. 2007).

It is important to understand the impacts of habitat management on the ecological components of habitats, but the provision of many ecosystem services also depends on the behaviour and psychology of humans (Gobster et al. 2007; Schaich et al. 2010a; Hernández-Morcillo et al. 2013). This is particularly true for cultural ecosystem services such as recreational experiences, which require a direct interaction between a visitor and a habitat (Chenoweth and Gobster 1990). The interdisciplinary nature of cultural ecosystem services makes them complex to analyse, and as a result their provision is under-researched (Rey Benayas et al. 2009; Hernández-Morcillo et al. 2013). Previous studies, particularly analyses of recreational experiences, have quantified people’s preferences for different components of habitats, in order to identify organisms or physical features that may improve recreational experiences (Adamowicz et al. 1998; Hanley et al. 1998). This approach measures the demand of visitors for habitat components, but it is limited as a predictive tool because it does not consider the ecological or behavioural mechanisms that underlie the supply of experiences to the visitor. To analyse recreational experiences more holistically it is necessary to understand the ecological and physical processes which affect the presence and distribution of habitat components (Haines-Young and Potschin 2010; Bastian et al. 2012), the interactions between visitors and habitat components (Chenoweth and Gobster 1990), and the relative preferences that people have for different components (Bullock et al. 1998; Dorwart et al. 2009). These three stages in the delivery of recreational experiences must be understood, in order to predict the net recreational experiences provided by a habitat, and the impacts of changes in management on recreational quality.

A detailed understanding of the provision of individual ecosystem services can inform their management, but an ecosystem service framework should also synthesise data on multiple services to inform the management of a whole suite (Haines-Young and Potschin 2010). An understanding of the trade-offs and synergies among a suite of ecosystem services can allow decision makers to predict the impacts of habitat management, and compare the suites of services that are likely to be provided under different management scenarios (Nelson et al. 2009). There is increasing interest in analysing trade-offs and synergies to understand how habitats can be managed to fulfil some of the objectives of an ecosystem service approach. For example, habitats could be managed to provide optimal suites of services (Lautenbach et al. 2010; Seppelt et al. 2013), or to deliver multifunctionality (Brandt 2003; Mander et al. 2007; Selman 2009; Haaland et al. 2011). It has been proposed that simulations of different habitat management scenarios could be used to investigate the range of ecosystem service outcomes that might be possible (Castellazzi et al. 2010; Lautenbach et al. 2010). Simulations have been successful in identifying optimal management strategies (Sanon et al. 2012), and in highlighting management options that could create multifunctional habitats (Rouquette et al. 2011). Previous studies have identified specific management regimes that could deliver these outcomes, but have not related optimality or multifunctionality to general characteristics of habitat management. To generalise the findings from case study simulations to other systems, patterns in habitat structure, such as habitat heterogeneity or connectivity, must be related to the characteristics of the suites of services that they provide. For example, it has been suggested that habitat heterogeneity may induce multifunctionality (Brandt 2003; Mander et al. 2007), but this has not been analysed through simulation.

An ecosystem service framework should provide decision makers with information about the expected impacts of habitat management on service provision (Nelson et al. 2009). However, little is known about the effects that ecosystem service information may have on decision making (Laurans et al. 2013; Laurans and Mermet in press). It is expensive and time consuming to collect and analyse ecosystem service information (Liu et al. 2010), so it is important to know whether decision makers are able to interpret and utilise such information. It is also important to predict the ways that ecosystem service information might change habitat management practices, to establish whether future habitat management is likely to deliver better protection for biodiversity, ecosystem service optimality, multifunctional habitats, or outcomes that satisfy the demands of a diverse stakeholder group. There are few well-documented case studies of applications of ecosystem service frameworks to habitat management problems (Naidoo and Adamowicz 2005) (though see (Fisher et al. 2008; Kushner et al. 2012)), but experimental decision making exercises can be used to analyse people's responses to ecosystem service information (Shaw 2012). Decision making processes can be complex because they commonly involve multiple people and organisations (Bakus 1982; Fish and Haines-Young 2011), but any decision is influenced by the preferences that the individual decision makers have for different management options (Bakus 1982). An individual's preferences for management options are relatively straightforward to measure, and the patterns of preferences within a group of people may give insights into the likely outcomes for management (Bakus 1982).

1.3. Floodplains as a study system for investigating the application of ecosystem service frameworks

Floodplains are the low-lying areas of land that surround rivers. In temperate regions natural floodplains are regularly or occasionaly inundated by water during periods of high flow, so are infrequently submerged (Tockner et al. 2000). As a result, floodplains are characterised by temporal and spatial hydrological heterogeneity (Junk et al. 1989; Benke et al. 2000; Amoros and Bornette 2002; Tockner et al. 2010). Hydrological heterogeneity provides a range of habitats which support a diverse range of organisms (Ward et al. 1999; Tockner and Stanford 2002). Floodplains can also provide a broad range of ecosystem services, such as flood regulation, clean water, agricultural production, and opportunities for recreational activities (Posthumus et al. 2010). Wetland areas, including floodplains, are one of the most important habitat types for providing ecosystem services to people (Costanza et al. 1997; Zedler and Kercher 2005).

Lowland river floodplains are one of the world's most heavily modified systems (Tockner and Stanford 2002), as many floodplains have been disconnected from regular flooding by flood defences and subsequently drained (Jungwirth et al. 2002); at least 90% of European floodplains have been modified in this way (Tockner and Stanford 2002. In the UK, only 42% of floodplains remain connected to their adjacent rivers, and only 4% of river reaches have adjacent habitat that is defined as "wetland" (Maltby et al. 2011). Floodplains have commonly been drained because they can provide fertile agricultural land, for example, two thirds of floodplains in the UK have been converted to grazed grasslands (Maltby et al. 2011). Around 80% of the population in temperate regions lives within 5 km of freshwater (Kummu et al. 2011), so many floodplains have become heavily urbanised (Baart et al. 2012). Past management of many floodplains has thus focused on two ecosystem services; the provision of food and the prevention of flood damage (Maltby et al. 2011) (Tockner and Stanford 2002; Baart et al. 2012). As a result, many of the other services provided by natural floodplains have been degraded (Tockner and Stanford 2002; Zedler and Kercher 2005; Maltby and Acreman 2011). The loss of some services, particularly biodiversity, has been noted, and this has stimulated debate about managing floodplains differently (Sparks 1995; Poff et al. 1997; Sparks et al. 1998; Jungwirth et al. 2002; Poff et al. 2003). More natural floodplain hydrology can be restored by removing or lowering flood defences, which reconnects rivers and floodplains hydrologically (Zsuffa and Bogardi 1995; Fischenich and Morrow 2000; Toogood et al. 2008), and an increasing number of such restoration projects are being proposed (Buijse et al. 2002; Tockner and Stanford 2002). The diversity of ecosystem services that floodplains can provide, and the conflicts surrounding their management, make them ideal for management under an ecosystem service framework (Rouquette et al. 2009; Posthumus et al. 2010).

Floodplains are a good model system in which to investigate the relationships between habitat management and ecosystem service provision because they are structured largely by one management factor; exposure to flooding (Morris et al. 2009; Rouquette et al. 2011). This relative simplicity means that the impacts of changes in hydrology are less likely to be confounded by changes in other forms of management. Changing the hydrological management of floodplains can cause extreme shifts in environmental conditions, and may even result in conversion from one broad habitat type (such as wet grassland) to another (such as open water). However, hydrological gradients in floodplains can also vary subtly (Jones et al. 2008), and flood inundation can be predictable and precisely controlled (Fischenich and Morrow 2000; Hammersmark et al. 2005). Floodplains therefore provide an opportunity to understand how ecosystem service provision varies across a continuous gradient of habitat management. Floodplain reconnection projects provide specific opportunities to analyse the effects of relatively large changes in habitat management regime.

Floodplain ecosystem services are also highly topical, as the recent UK National Ecosystem Assessment has drawn attention to the diversity of services that freshwater ecosystems, including floodplains, can provide (Maltby et al. 2011). Many of the services provided by freshwater habitats have been inadequately recognised in the past, and as a consequence, there are no remaining floodplains that have not been altered to some extent by human activity (Maltby et al. 2011). The freshwater chapter of the National Ecosystem Assessment recommended that restoration of floodplains is necessary to maintain and improve the provision of ecosystem services, but noted that there is considerable uncertainty about the relationships between habitat condition and service provision (Maltby et al. 2011). Uncertainty about the relationships between hydrology, ecology, and ecosystem service provision makes it difficult to predict the specific outcomes of changes in floodplain management. This uncertainty may explain the shortage of specific objectives relating to floodplain restoration or creation in the UK Governmental White Paper (DEFRA 2011), which responded to many of the concerns raised by the National Ecosystem Assessment. The Natural Choice White Paper recognised the multiple benefits provided by freshwater ecosystems and the importance of restoring more natural freshwater systems (DEFRA 2011). Wetlands were mentioned due to their utility as sustainable drainage systems, but few mentions were made of floodplain wetlands in particular, and no specific objectives or funding announcements were made for wetland restoration (DEFRA 2011). In comparison, specific objectives (and in some cases funding) was announced for other aspects of freshwater ecology, such as river water quality, catchment sensitive farming, and water abstraction licensing (DEFRA 2011). A greater understanding of floodplain ecosystem services and the impacts of restoration is required to increase confidence in floodplain restoration as a strategy, which may lead to its greater implementation.

1.4. Objectives of this study

This study aims to inform the implementation of ecosystem services as a framework for habitat management, with a focus on the management of floodplains at a local scale (ca. 55 ha) across a moderate gradient of environmental manipulation (the transition between permanent standing water and infrequent flood inundation). It focuses on the four related knowledge gaps that were identified in section 1.2., with four specific objectives; to (1) provide a better understanding of the ecological impacts of increasing floodplain hydrological connectivity, (2) provide a framework for analysing the quality of recreational experiences at small spatial scales, (3) explore the effects of floodplain hydrological heterogeneity on ecosystem service optimality and habitat multifunctionality, and (4) investigate the effects of ecosystem service information at different levels of detail on people’s preferences for floodplain management scenarios. This study addresses these aims through five data chapters which are written as separate manuscripts. An additional chapter introduces and provides some background to the case study floodplain. The final chapter synthesises the findings of the study as a whole, and discusses the implications for the management of floodplains, and the application of ecosystem services as a framework for managing habitats more generally.

Chapter 2 of this study provides an introduction to the study floodplain. The Fishlake floodplain has a history of channelisation and drainage, but has more recently been reconnected hydrologically to the adjacent river. This chapter describes the location and history of the site, and then goes into further detail about the restoration work and its effects on topography and hydrology. Chapter 2 also introduces a key hydrological dataset that is used in many of the following chapters.

Chapters 3 and 4 of this study analyse the effects of hydrological management on two ecological taxa that are important in providing ecosystem services. Chapter 3 considers the composition of plant communities, which are of interest for biodiversity conservation at both Local Authority and National scales. It analyses the drivers of changes in plant community composition following hydrological restoration, and reports the trends observed in the study floodplain over a period of four years. Chapter 3 also discusses the trajectories of plant community change and the constraints on change, in relation to the targets set by the restoration project. Chapter 4 considers the impacts of floodplain hydrology on European water voles (*Arvicola terrestris*), which are a popular and charismatic species of high conservation interest in the UK (Strachan et al. 2011). It describes the habitat preferences of water voles in a floodplain wetland, analyses the resilience of water voles to flood events, and discusses the potential for the hydrological restoration of floodplains as a conservation tool for this species.

In Chapter 5, floodplain recreational experiences are analysed to allow spatial variation in the quality of experiences to be predicted. The net quality of the experience provided by odonates (dragonflies and damselflies, which can have positive impacts on people's recreational experiences) and debris items (which are commonly perceived negatively) is investigated. Chapter 5 combines three pieces of analysis that describe; the relationships between ecological and physical factors and the distribution of odonates and debris, the interactions between visitors and these habitat components, and the relative preferences that people have for different habitat components. These three components are combined to form a spatially-explicit model that predicts the relative quality that a visitor would experience, at any location in the study floodplain. The recreational experience model is subjected to sensitivity analysis to evaluate the relative importance of the three stages in contributing to the accuracy of the estimates. Finally, the model is applied to a hypothetical management question to inform the design of wildlife viewing site locations.

Chapter 6 investigates the management of a suite of ecosystem services, and utilises some of the information provided in the previous chapters. This chapter first analyses the provision of seven ecosystem service indicators across 100 quadrats that make up a hydrological gradient, from open water to wet grassland. It then uses simulation to explore the range of hypothetical floodplain mosaics that could be created if different survey quadrats were combined to make up larger mosaics. This simulation is optimised to identify the optimal management strategies. The optimal simulated floodplain mosaics are then assessed for their multifunctionality. These two characteristics of the suite of services provided are then analysed in relation to the heterogeneity of the floodplain mosaics, to identify trends in management that could provide optimal suites of services or multifunctional floodplains.

Chapter 7 assesses the use of ecosystem service data in habitat management decision making. This chapter synthesises information from the preceding chapters to generate seven hypothetical floodplain management scenarios. This chapter analyses the effects that different levels of ecosystem service information can have on people's preferences for the different habitat management options. The floodplain management scenarios are described to non-expert participants in different amounts of detail. Differences in people's preferences depending on the quality of the information that they are provided with are discussed in relation to the potential impacts for individual and group decision making.

Chapter 8 reviews the principal findings of the data chapters and discusses the implications for floodplain management; in particular the constraints to restoring floodplain ecology, and the role that floodplains can play in the wider landscape. It then discusses the implications for future applications of ecosystem services, such as the practical issues that may face organisations that wish to conduct assessments, and the issue of uncertainty about ecosystem service outcomes.

To provide a reference for the various datasets used within this thesis, a summary of the key datasets, and the chapters in which they are utilised, can be found in Table 1.1.

Table 1.1. Summary of datasets used in the separate chapters of this thesis, the number of spatial replicates (i.e. quadrats) and the temporal characteristics of the surveys.

Dataset	Chapters that utilise these data	Number of spatial replicates	Survey dates
Plant community composition in sample plots	3	28	June 2009, 2011, 2012, and 2013
Water vole distribution (hand search for field signs)	4	NA	September – October in 2008 and 2011.
Water vole presence and absence at artificial rafts	4	34	September – October in 2011 and 2012.
Presence and abundance of debris items in floodplain quadrats	5,6	100	June 2012, August 2012 and May 2013
Presence and abundance of odonates in floodplain quadrats	5,6	100	Three days in August 2012 and three days during July-August 2013
Plant community composition in floodplain quadrats	6	100	June – July 2012
Presence of cattle dung in floodplain quadrats	6	100	Once in July 2012 and once in April 2013
Presence of wetland birds in floodplain quadrats	6	100	Six occasions in June 2012, and six occasions between May and June 2013

Table 1.1 Continued.

Dataset	Chapters that utilise these data	Number of spatial replicates	Survey dates
Topographic surveying	2,3,4,5,6	NA – continuously mapped over site	Surveyed by Environment Agency LIDAR in 2008. Amended following total station surveys in 2011 and 2012.
Flood exposure mapping in the floodplain	2,3,4,5,6	NA – continuously mapped over site	Every two weeks between 04/05/2011 and 19/04/2013
Water level monitoring in the River Don	2,3	1	Every 10 minutes between the 28/01/2009 and the 31/07/2013

2. Fishlake floodplain restoration and hydrological modelling

The study described in this thesis investigated the implementation of ecosystem services as a framework for the habitat management of floodplains. This study focused on a case study floodplain that had recently been hydrologically reconnected to a river. Case study approaches can be limited in the generality of their conclusions, because the variability of floodplain responses is not measured, so it is not clear whether consistent conclusions would be drawn at other floodplain restoration projects (Eisenhardt 1989). However, there were no similar, recently restored floodplain sites with comparable baseline data in the vicinity of the study site, so replication was not possible. By focusing on a single case study it was possible to increase the level of detail of an analysis, for example by considering a larger number of ecosystem services and analysing the target services more precisely. The Fishlake study site is likely to differ from other floodplains in terms of the habitat types that are present and the ecosystem services that are provided. However, as a historically drained floodplain that was used for agriculture and flood storage prior to restoration, the key pressures on Fishlake were those that have driven trends in historical floodplain management across Europe.

This chapter provides an introduction to the study system that is used throughout the rest of the thesis. Section 1 describes the location, history, and recent management of the site, and discusses some ecosystem services that the floodplain provides. Section 2 gives an overview of the Fishlake floodplain restoration project, which began in 2009. This section describes the objectives of the project, the engineering work that was conducted, and the impacts of this engineering on the hydrology of the floodplain. Section 3 summarises the research opportunity provided by the Fishlake floodplain restoration, and introduces some key datasets that are used in later chapters.

2.1. Background to the Fishlake floodplain

2.1.1. Overview

The Fishlake floodplain is located between the settlements of Fishlake, Thorne, and Stainforth (Latitude: 53.611239, Longitude: -1.002889), in the Metropolitan Borough of Doncaster, in the United Kingdom (Figure 2.1a). The site is a washland of the River Don (Hiley et al. 2008) and is located within the Humberhead Levels (Figure 2.1b). The Don drains much of South Yorkshire (Faulkner and Wass, 2005) and is a major river; the mean discharge at Doncaster, which is approximately 11 km upstream, is 16 $m^3 s^{-1}$ (CEH 2013). The river level at Fishlake is tidally influenced with a reach of approximately 3.5 m during a spring tide, and 1 m during a neap tide (Hiley et al. 2008). The tidal reach is caused by freshwater backing up from the estuary, so is not saline (Hiley et al. 2008). The Fishlake floodplain extends on both banks of the Don, over an area of 25 hectares on the north bank, and 22 hectares on the south bank (Figure 2.2).

2.1.2. History of the Fishlake wetlands and the wider region

Approximately 20% of the land area of England drains into the Humber Estuary, and much of the water passes through the Humberhead Levels (Van de Noort, 2000). The Levels were historically the third largest fenland in England (Rotherham and Harrison, 2006), and provided a mosaic of wetland habitat types, including open water, marsh and wet woodland (Rotherham and Harrison, 2006). Economic activity in the Fishlake area relied heavily on the local wetlands for resource extraction, and Fishlake village was a key trading centre on the Don (Gaunt, 2012). The region was home to major freshwater fisheries such as the Hatfield Chase, and also produced quantities of venison and waterfowl (Rotherham and Harrison, 2006). Other extractive industries were reed production and peat cutting (Rotherham and Harrison, 2006).

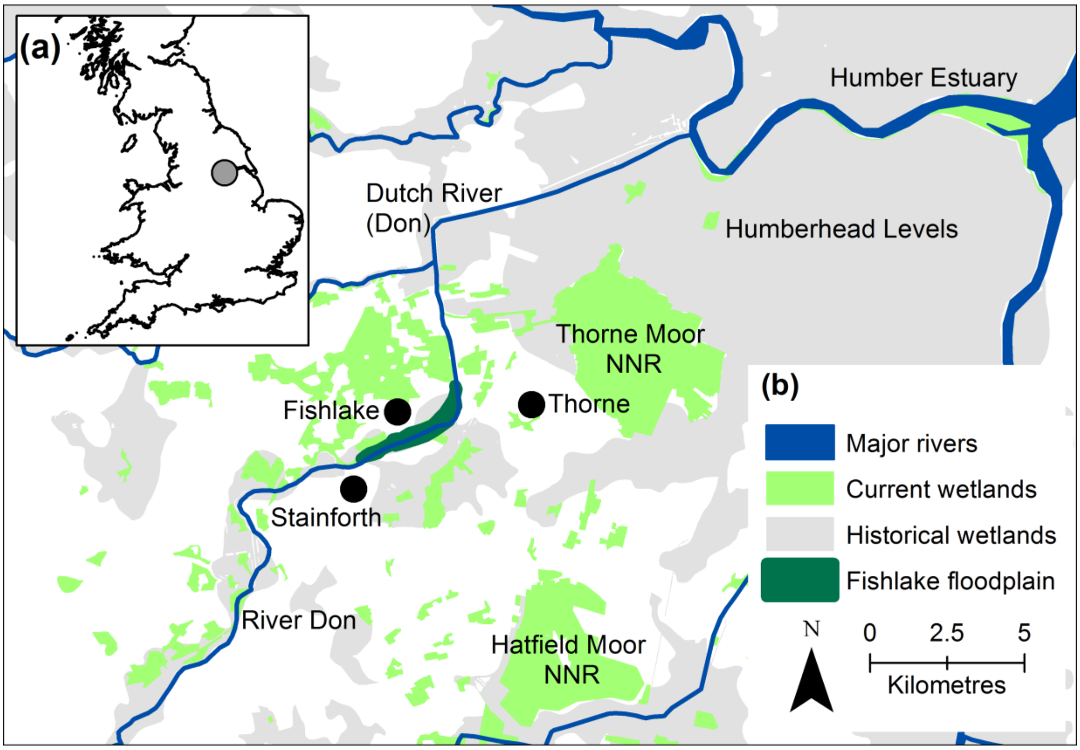

Figure 2.1. (a) Location of the Fishlake floodplain in the UK. (b) Location of the Fishlake floodplain in the Humberhead Levels. The estimated historical and current extent of wetlands in the region is also indicated, based on data provided by the Wetland Vision (Hume 2008).

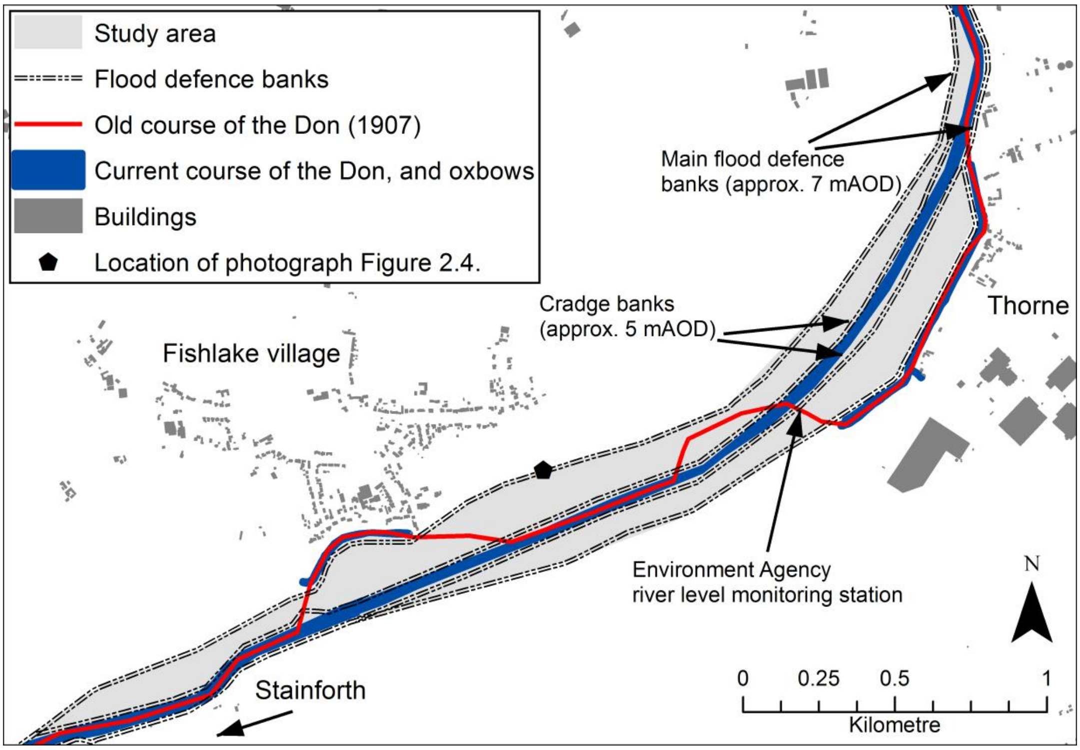

Figure 2.2. The Fishlake floodplain and its location in relation to local settlements. The pre-straightening course of the Don, as well as the current course and flood defences, are also shown.

Many of the wetlands in the Fishlake area were drained following a deal between King Charles I and a Dutch consortia led by the engineer Cornelius Vermuyden in 1627 (Gaunt, 2012). This led to a major re-routing of the River Don through the “Dutch River” (Figure 2.1b), and the subsequent loss of many wetlands around Fishlake and Thorne (Gaunt, 2012). The area of wetlands declined further over the following centuries as agriculture and coal mining became the most important local industries (British Coal 1989; Eversham and Skidmore, 1991; Gearey et al. 2009).

The Don was straightened at Fishlake in 1947 as part of a larger flood protection programme carried out by the River Ouse Catchment Board (Trudgill 2007; Figure 2.2). As part of this engineering the river was also prevented from flooding by earth flood defence banks. The Fishlake floodplain is unusual in having two lines of flood defence; there are low (approximately 5 m above sea level) cradge banks on the riverbank, with more substantial main banks (which are approximately 7 m above sea level) behind (Figure 2.2). The area between the flood banks has developed as an infrequently flooded washland (Hiley et al. 2008).This area, referred to as the Fishlake floodplain or Fishlake wetland, is the study area for the work described in this thesis (Figure 2.2).

2.1.3. Recent management of the Fishlake floodplain and some key ecosystem services

The Fishlake floodplain has been owned by the UK Environment Agency (or its precursors) since the 1940s, and has been managed primarily as a flood storage area (Hiley et a. 2008). Fishlake was one of a number of areas that were designed to flood at high river flows, to relieve pressure downstream on the river network (Hiley et al. 2008). The exact importance of the Fishlake floodplain as a flood storage area is not known, although the capacity of the site is relatively small compared to the large volumes carried in the Don (Faulkner and Wass, 2005). However, Fishlake is clearly regarded as a key part of the flood regulation network by the Environment Agency, as maintaining the provision of flood storage was a major concern during the planning of the floodplain restoration project which forms the basis of this study (see Section 2.2.; Hiley et al. 2008). Furthermore, flood regulation is a major issue for local residents; 65% of 83 Fishlake residents reported flooding as their greatest crime or safety-related concern (Fishlake Village Plan 2010).

There is evidence in the First Land Utilisation Survey of Great Britain that the Fishlake floodplain has been grazed by cattle and ponies since before the river was straightened in the 1940s (Stamp 1935). A herd of around 100 beef cattle are present at the site between March and October every year (Figure 2.3). The herd belongs to a tenant farmer and is made up mainly of breeding cows and calves, with a number of older bullocks and heifers, and two adult bulls. The floodplain also provides additional value to the farmer, as it has been managed under a Natural England Stewardship scheme since 2011. The combined value of the Entry Level and Higher Level scheme is £39,700 per year (Natural England, 2013). The wetlands are commonly used for unauthorised grazing; at least 10 cob ponies are consistently present during the summer, with more during the winter (the maximum recorded was 54; Figure 2.3).

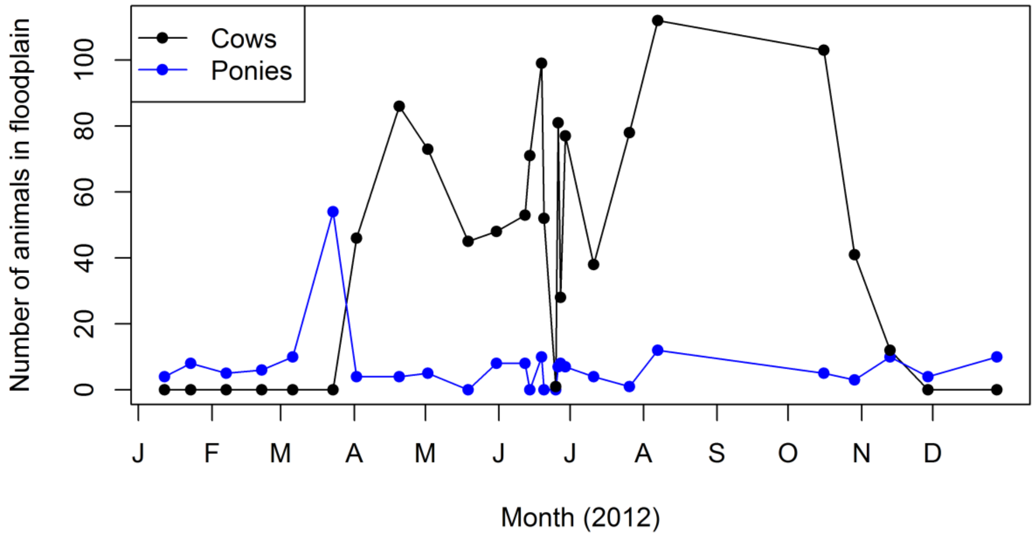

Figure 2.3. Cow and pony numbers in the Fishlake floodplain on 27 survey visits during 2012. Animals were only counted if they were within the main flood defence banks.

Several local walking routes pass through the floodplain area (Ordnance Survey 2006), and the main flood defence bank is commonly used as a footpath (Environment Agency 2009). A survey of people in Thorne in October 2013 reported that 11 of 47 participants (23%) had ever visited visited the Fishlake floodplain, and of these, 8 had visited for recreational walking (Lowe 2014). Visitors are generally local people from Fishlake or Thorne who are walking dogs or otherwise passing through (John Deeney, local resident; personal communication), and visitor numbers are low (the population of the closest settlement, Fishlake, is 628 (ONS 2001). A local walking group who are based in Thorne occasionally walk a circuit of the floodplain (approximately 30 visitors, encountered twice over at least 80 site visits over three years). Despite the low observed use of the floodplain for recreation, the appearance of the river and the footpath route along the main flood defence bank is “very important” to over 70% of Fishlake residents (Fishlake Village Plan, 2010).

A small number of local anglers (between 5 and 10) infrequently use the floodplain to access the River Don, most commonly adjacent to the bridge at Stainforth, at the western end of the wetland, or directly parallel to Fishlake village on the north bank (John Deeney, local resident; personal communication). The wetland areas within the floodplain are not suitable for angling; a fry survey by the Environment Agency in 2011 found only 3 individuals of fishable species at 10 drift netting sites (Environment Agency 2011). The wetland on the southern bank of the site is targeted infrequently by duck hunters.

There are a variety of habitat types present at the Fishlake floodplain. The floodplain supports a range of aquatic, marsh, and wet grassland plant communities (Hiley et al. 2008; Shaw et al. 2009; Figure 2.4). A number of duck, swan, and wading bird species use the floodplain throughout the year (Hiley et al. 2008). European water voles, which are a threatened species in the UK, are present both in the surrounding drainage ditch network and in the floodplain itself (Hiley et al. 2008). There is some limited wildlife viewing activity at the site; one local resident has observed the bird community almost daily for several years (John Deeney, personal communication). A community blog for bird enthusiasts was active during the spring of 2011 but has since been dormant (Deeney 2011). Local wildlife organisations such as the Yorkshire Wildlife Trust, Yorkshire Naturalists, and Thorne and Hatfield Moors Conservation Forum do not visit the site for recreational, educational, or monitoring purposes.

Figure 2.4. Photographs of the Fishlake floodplain, taken from the point marked on Figure 2.2. (a) The view south from the main flood bank, towards the River Don. This part of the floodplain contains open water, wet grassland and marsh habitats. (b) Close-up of marsh vegetation; mainly spike rush (*Eleocharis palustris*) with some reed sweetgrass (*Glyceria maxima*). Both photographs were taken on the 03/05/2009.

2.2. The Fishlake floodplain restoration project

2.2.1. Objectives of the Fishlake restoration project

In 2008 an opportunity to create and improve habitats in the Fishlake floodplain came about as part of a wider flood regulation and defence improvement programme (Hiley et al. 2008; Environment Agency 2009). As many floodplains have been degraded by disconnection from a natural flooding regime, restoration of hydrological connectivity between rivers and floodplains is intended to restore natural habitats and provide opportunities for wetland species (Buijse et al. 2002). Such habitat improvement was a policy objective of the Environment Agency (Hiley et al. 2008), and the unusual cradge bank flood defences at Fishlake provided a controlled area in which hydrological connectivity between the river and floodplain could be restored without compromising the main flood defences (Environment Agency 2009). The Fishlake floodplain restoration project was not explicitly designed using an ecosystem services framework, but a number of ecosystem services (while not termed as such at the time) were considered, and a range of stakeholder groups were consulted (Environment Agency 2009). The Environment Agency had five main objectives when designing the restoration strategy (summarised from Hiley et al. 2008; Environment Agency 2009);

(1) To create and maintain areas of biodiversity action plan (BAP) habitats, such as floodplain grazing marshes, and to a lesser extent marsh and swamp.

(2) To provide areas of suitable habitat for key taxa such as wetland birds, water voles, and riverine fish, and ensure that engineering work and restored flooding would not negatively affect these taxa.

(3) To improve and encourage public access to the floodplain.

(4) To maintain the same level of flood protection for the local area, and to maintain some flood storage capacity.

(5) To maintain a similar level of livestock grazing to support the livelihood of the tenant farmer.

Some of these objectives, such as creating BAP habitat and providing habitat suitable for wetland birds, require similar management and so could largely be achieved together. However, there were also trade-offs between different objectives; increasing the normal water level in a floodplain could be beneficial for wetland birds and fish (Mauchamp et al. 2002), but would reduce the flood storage capacity, and could reduce the area available for grazing cattle (Morris and Brewin in press). The final restoration design therefore aimed to achieve a satisfactory balance of the five objectives.

2.2.2. Restoration engineering and other actions

The final restoration design aimed to increase the frequency of flood events from between zero and twelve times a year, to at least thirty-six times a year (Hiley et al. 2008). The cradge banks were lowered at two locations on the northern bank in August 2009 (Figure 2.5), which allowed flooding to occur at lower river levels (Hiley et al. 2008). At each location an eight metre section of the cradge bank was lowered by approximately 1.5 m (from 5.2 m to 3.7 m) and reinforced with concrete cladding to prevent erosion (Figure 2.6). It was expected that regular flooding would be driven mainly by spring tides (Hiley et al. 2008), and flood water was expected to drain from the floodplain relatively quickly after river levels dropped, due to the addition of valved culverts that allowed water to flow out of the floodplain but not in (Hiley et al. 2008). Rapid drainage was expected to provide suitable hydrological conditions for floodplain grazing marsh plant species (Hiley et al. 2008), and was also considered important for the maintenance of flood regulation, as pooled standing water would reduce the available capacity (Hiley et al. 2008).

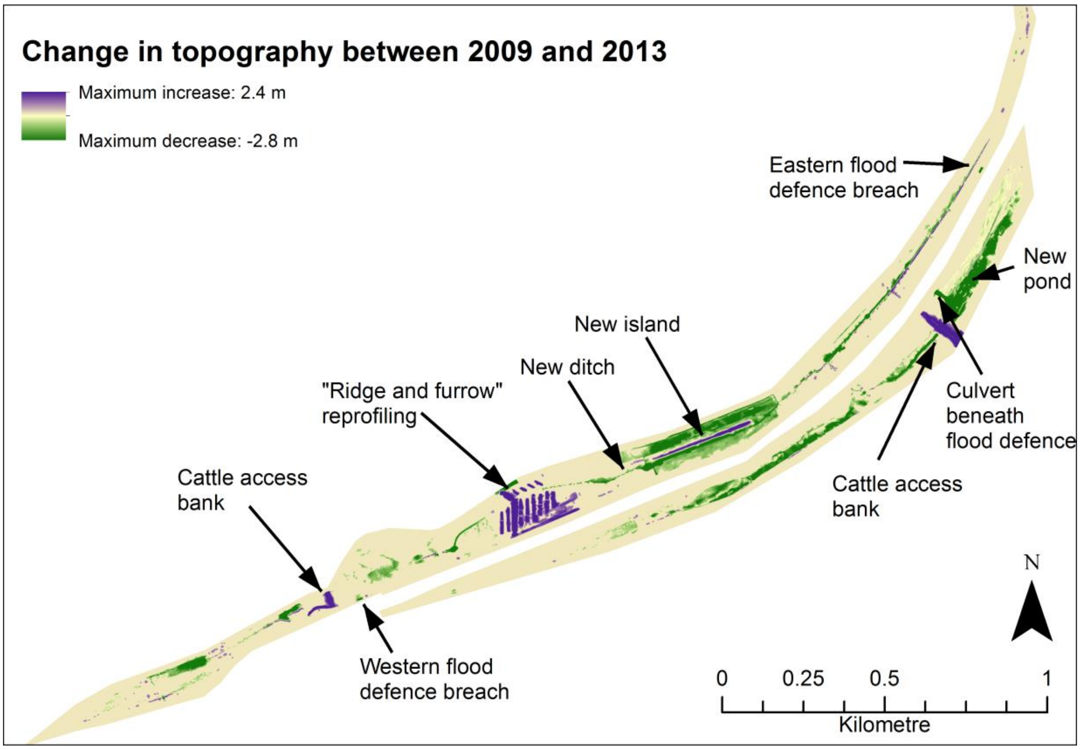

Figure 2.5. Changes in topographic height of the Fishlake floodplain between 2009 and 2013. A high-resolution (1 pixel = 0.0625 m^2) LIDAR survey was carried out by the Environment Agency prior to restoration. A post-restoration elevation model was then created using a Leica TCRP1205 robotic theodolite and electronic distance meter, between 2011 and 2013. The two elevation models were georeferenced and normalised using fixed reference points. The 2013 elevation was then subtracted from the 2009 elevation to show the net change in elevation over this period.

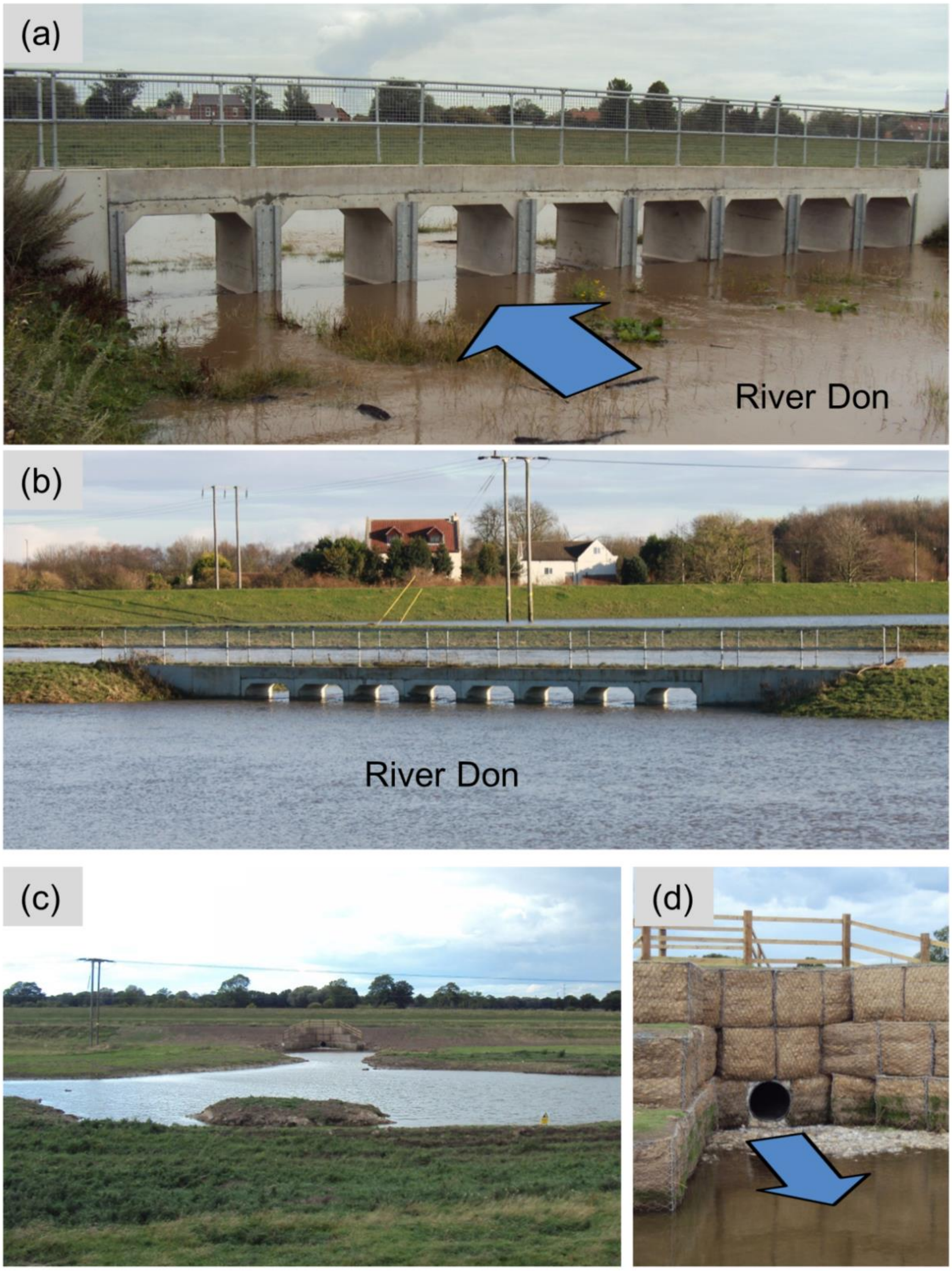

Figure 2.6. Engineering work carried out at the Fishlake floodplain. (a) Concrete structure put in place at western bank breach on the north bank of the river, as seen from the river channel. Water is passing into the floodplain from the river, as indicated by the blue arrow. Taken on the 31/08/2011. (b) The western bank breach seen from the far bank of the river, at a higher state of flood. Note the high level of flood water within the floodplain. Taken on the 29/11/2012. (c) New pond dug on the south bank of the river, as seen from the main flood defence bank, looking north. (d) close up of culvert structure shown in (c). Water can flow into the floodplain through the culvert, as indicated by the blue arrow. Both (c) and (d) were taken on the 30/09/12.

A number of existing ditches at the site were dredged and enlarged during June-August on 2009, and in some parts of the site new drainage ditches and pools were dug (Figure 2.5) to create additional habitat for wetland birds, fish and water voles (Hiley et al. 2008). To maintain cattle and public access during flood conditions a number of raised banks were added to connect the cradge banks to the main bank (Figure 2.5). A "ridge and furrow" pattern of raised banks was added to increase topographic variation and provide refuges for wetland birds during flood events (Environment Agency 2009).

The floodplain on the southern bank of the Don was connected hydrologically to the river via a culvert in September 2011 (Figure 2.5). This culvert was designed to fill a large, newly dug pond to provide spawning habitat for fish fry (Environment Agency 2009; Figure 2.6), and was topographically lower than the cradge bank breaches to maintain more consistent hydrological connectivity. The new pond was designed to feed water in to the other water bodies on the southern side of the floodplain through a culvert and ditch network (Environment Agency 2009).

Apart from the hard engineering to increase hydrological connectivity and reprofile the topography, few additional restoration measures were implemented. Some reprofiled areas were seeded with a sward mix containing species common to the site (including white clover *Trifolium repens* and perennial rye-grass *Lolium perenne*) (Richard Jennings, Environment Agency; personal communication). The tenant farmer entered into a Natural England Higher Level Stewardship scheme in May 2011, under which he was obliged to maintain cattle grazing at a similar level to the previous herd size. There were additional obligations to annually remove ragwort (*Senecio jacobea*) and thistle (mainly *Cirsium arvense*), and to clear vegetation around the drainage ditch network (Natural England, 2013). These management obligations were previously part of the tenancy agreement or were conducted directly by the Environment Agency (Hiley et al. 2008), so these aspects of management did not substantially change following the hydrological restoration. A number of public access improvements, such as the conversion of stiles to swing gates, were carried out in 2012.

2.2.3. Hydrological effects of floodplain reconnection

The detailed baseline topographic dataset allows the engineering of the Fishlake restoration project to be accurately quantified. However, the hydrological impact of topographic changes is more complex to analyse, as no on-site hydrological monitoring was carried out prior to hydrological reconnection. In this section (2.2.3) the hydrological impacts of the Fishlake restoration project are estimated using two hydrological datasets. First, topographic and river level data were analysed to estimate the increase in frequency and duration of flood events due to the cradge bank lowering. Second, a statistical hydrological model was created to link the frequency of flood events to flood exposure in different parts of the floodplain.

2.2.3.1. Hydrological modelling methods

The number and duration of flood events were analysed by combining topographic measurements of the minimum flood entry point with river level data from an Environment Agency monitoring station located in the centre of the site (Figure 2.2). The river level was monitored at 10 minute intervals between the 28/01/2009 and the 31/07/2013. The floodplain was compartmentalised into 12 topographically separate hydrologic sections that were assumed to act as distinct drainage basins (Figure 2.7). The lowest point of the flood defences was found for each hydrologically isolated section of the floodplain (Figure 2.7); this was either the lowest part of the cradge bank or the height of a culvert that was known to allow water to flow into the floodplain (see Appendix A.1 for a table of the lowest points in each section every year). Flooding was identified as any time point when the river level was higher than the flood entry point, and the duration of flood events was defined as the number of consecutive time points when flooding was occurring. This process was carried out twice for all flood entry points; once under the 2009 topography, and once under the 2013 topography. For brevity, the effect of lowering the flood entry point on the flood frequency and duration of flood events was directly compared for only one flood entry point; the westernmost cradge bank breach. The results for the other entry points were not directly compared but were used in the next stage of the hydrological modelling procedure.

To link the frequency of flood events to hydrological conditions within the floodplain, hydrological records within the floodplain were required. The surface area of standing water in the floodplain was recorded at approximately two week intervals for two years, between 04/05/2011 and 19/04/2013. On each sampling occasion, the boundaries of each water body were walked and were recorded at approximately 15-step intervals using a Garmin Oregon 450 global positioning system (GPS). These points were then cross-referenced with a high-resolution topographic model. The topographic model was temporally explicit, so accounted for the construction of the new pond in the south east of the floodplain in October 2011. The cross-referencing process generated a number of estimates of the height of the water level in the floodplain. The water level within each hydrologically distinct section of the floodplain was calculated as the mean of the estimates that were recorded within it. The areas of the floodplain that were topographically lower than the estimated water level were then classified as submerged. This monitoring programme allowed the submerged area of the floodplain to be mapped on 52 sampling occasions at a high spatial resolution (1 pixel = 0.0625 m^2). The proportion of the 52 sampling dates that a location was submerged (proportional submergence) was used as the measure of relative wetness.

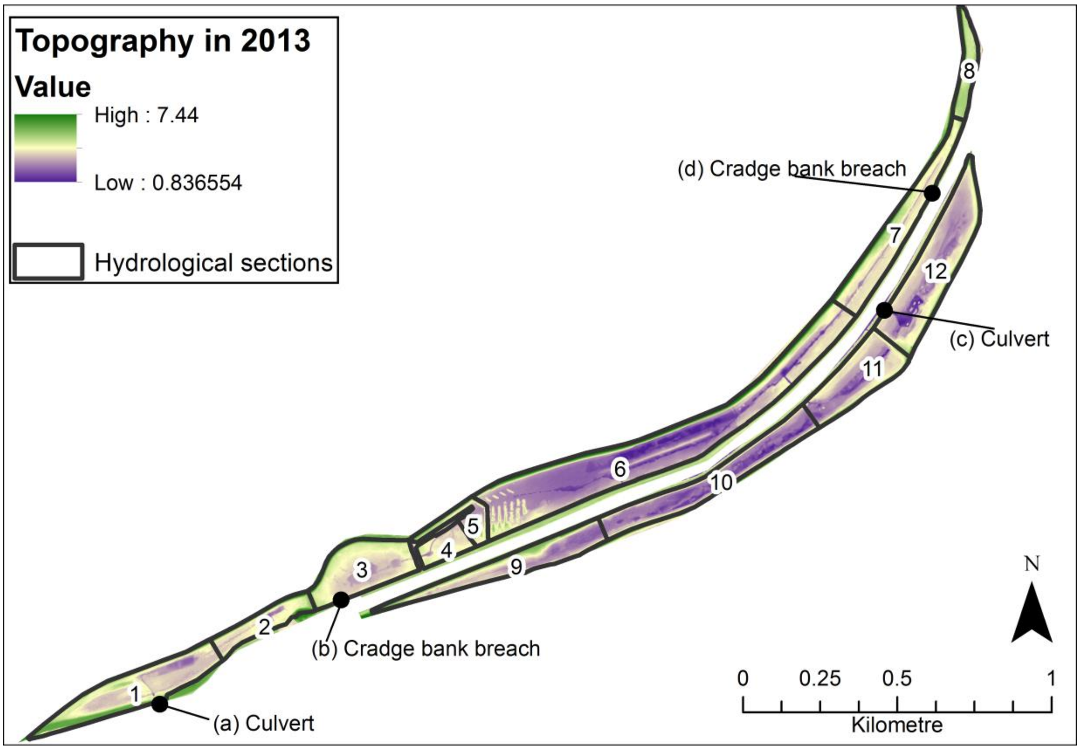

Figure 2.7. Topography of the Fishlake floodplain, and hydrological sections used for water level sampling and hydrological modelling. Hydrological sections 1 and 2 are flooded from flood entry point (a). Section 3 is flooded through point (b). Sections 4, 5, 6, 7, and 8 are flooded through point (d). Sections 9, 10, 11 and 12 are flooded through point (c).

A uniform grid of 1992 points was generated across the floodplain area, and hydrological conditions were modelled independently at each grid point. To create a predictive model of floodplain hydrology, the proportional submergence data were linked to the river level data using logistic regression. The probability that a location on the floodplain will be submerged at any point in time depends to some extent on the amount of recent hydrological connectivity to the river. A separate logistic regression model was constructed for each grid point, and submergence of the point on each of the 52 sampling occasions was modelled as a binary variable (either submerged or not submerged on each occasion). The proportion of the preceding two weeks that the river water level was higher than the lowest point of the flood defence (the proportion of time when flooding was occurring) was used as the explanatory variable. The resulting logistic regression models were then used to predict the proportional submergence of each location from a time series of flood events, over the period of field sampling (between 04/05/2011 and 19/04/2013). Proportional submergence was predicted twice for each location under different conditions; first using flood frequencies calculated using the baseline topography, and second using flood frequencies calculated under the restored (post-hydrological reconnection) topographic conditions.

The statistical hydrological model assumed that the topographic height of each location did not change, so was only applicable in areas that were not altered topographically as part of the restoration. Locations that changed in height between the topographic surveys in 2009 and 2013 (Figure 2.7) by more than one standard deviation from the mean were therefore excluded from subsequent analyses (565 locations were excluded). Standard statistical methods were not appropriate when analysing the difference in proportional submergence predicted by the baseline and restored hydrological models because the sample size (the number of grid points), and therefore the number of degrees of freedom available, was arbitrary. The comparison was made simply as the mean of the differences between the baseline and restored predictions at the 1492 locations. To assess the accuracy of the statistical hydrological model, the predicted post-restoration submergence events were compared to the sample data.

2.2.3.2. Results of hydrological modelling

The river level at Fishlake was variable due to the strong daily and seasonal tidal influence (Figure 2.8). At a cradge bank height of 5.3 m, the floodplain would have experienced eight flood events over the study period, an average of 1.7 every year. The median duration of flood events would have been 16.5 hours, and the longest flood would have been approximately 2 days. Under the post-restoration topography, with a breach height of 3.7 m, an estimated 571 flood events would have occurred over the study period. This is an average of 11 flood events every month, although the frequency changed seasonally. The median duration of a flood event would have been 50 minutes, and the longest flood event would have lasted for approximately five days.

Overall, the statistical hydrological model correctly predicted submergence, when compared to the monitored data, in 87% of cases. Submerged cases were correctly predicted 90% of the time, while unsubmerged cases were correctly predicted 75% of the time. This suggests that the hydrological model slightly over-estimated the probability of locations being flooded. The effect of this estimation should be consistent between the 2009 and 2013 models, so comparing the predictions from these models should still allow relative differences in wetness to be assessed. The difference in proportional submergence predicted by the 2009 and 2013 models varied spatially, with the southern bank of the floodplain in particular showing a large increase in the proportion of time that it was submerged (Figure 2.9). On average the reconnected floodplain was predicted to be submerged for an additional 10% of the period, compared to the baseline prediction.

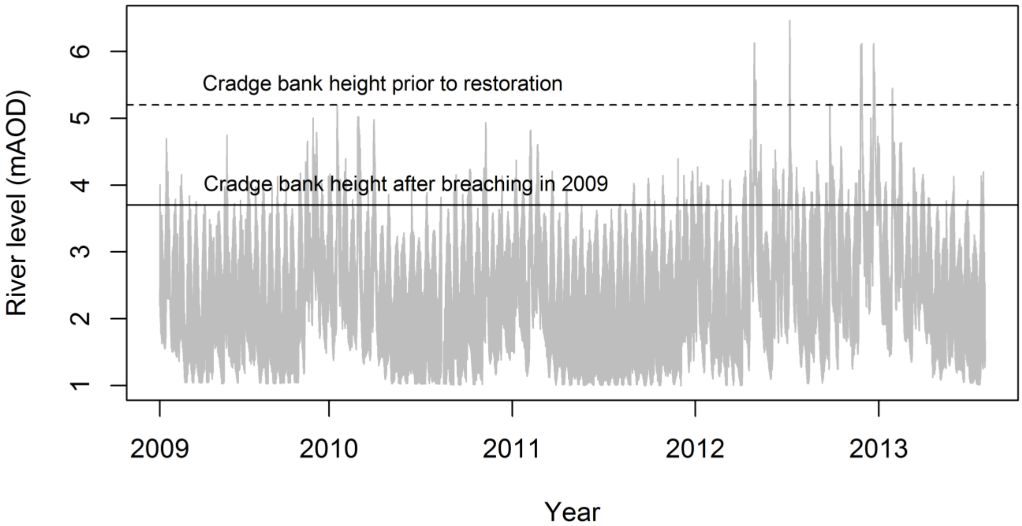

Figure 2.8. Time series of river level data between 28/01/2009 and 31/07/2013. River level is measured in metres above the ordnance datum at Newlyn, in Cornwall, which is approximately sea level.

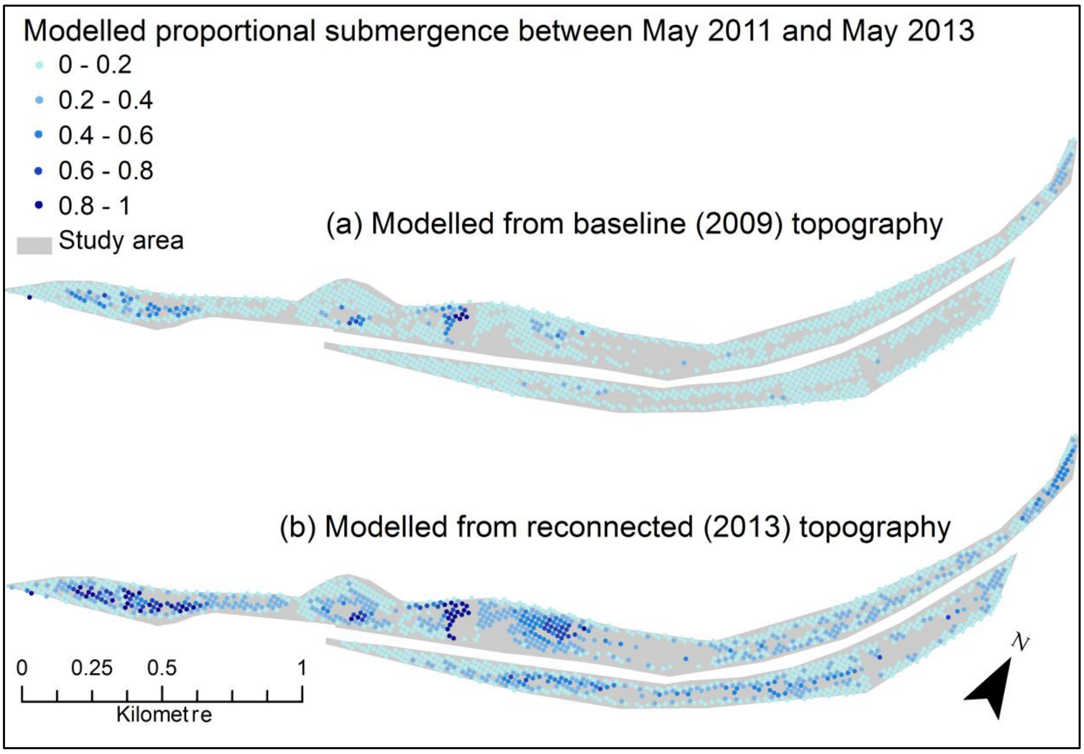

Figure 2.9. Proportional submergence of the Fishlake floodplain modelled under two flooding scenarios. Submergence was modelled from a time series of river level data between May 2011 and May 2013.

2.2.3.3. Summary: the hydrological impacts of restoration

The Fishlake restoration project has been successful in increasing the degree of hydrological connectivity between the river and the floodplain. Such lateral hydrological connectivity can transport materials and organisms, and can thus have important ecological implications (Tockner et al. 1999; Amoros and Bornette, 2002). Furthermore, the increased degree of flood connectivity altered the hydrological characteristics of the habitats within the floodplain. While the hydrological models slightly overestimated the probability of some areas of the floodplain being submerged, a substantial increase in the proportion of the time that the floodplain was submerged was predicted following the reconnection. The floodplain reconnection created not only wetter floodplain habitats, but also a variety of different habitat types (Figure 2.10). As planned, water appears to drain rapidly from the floodplain when river water levels recede, and there are large areas of infrequently submerged habitats (Figure 2.10) that should be suitable for cattle grazing (Morris and Brewin in press) and development into floodplain grazing marsh (Wheeler et al. 2004; Hiley et al. 2008). Example photographs showing the difference between high and low water levels in the floodplain can be seen in Figures 2.11 and 2.12.

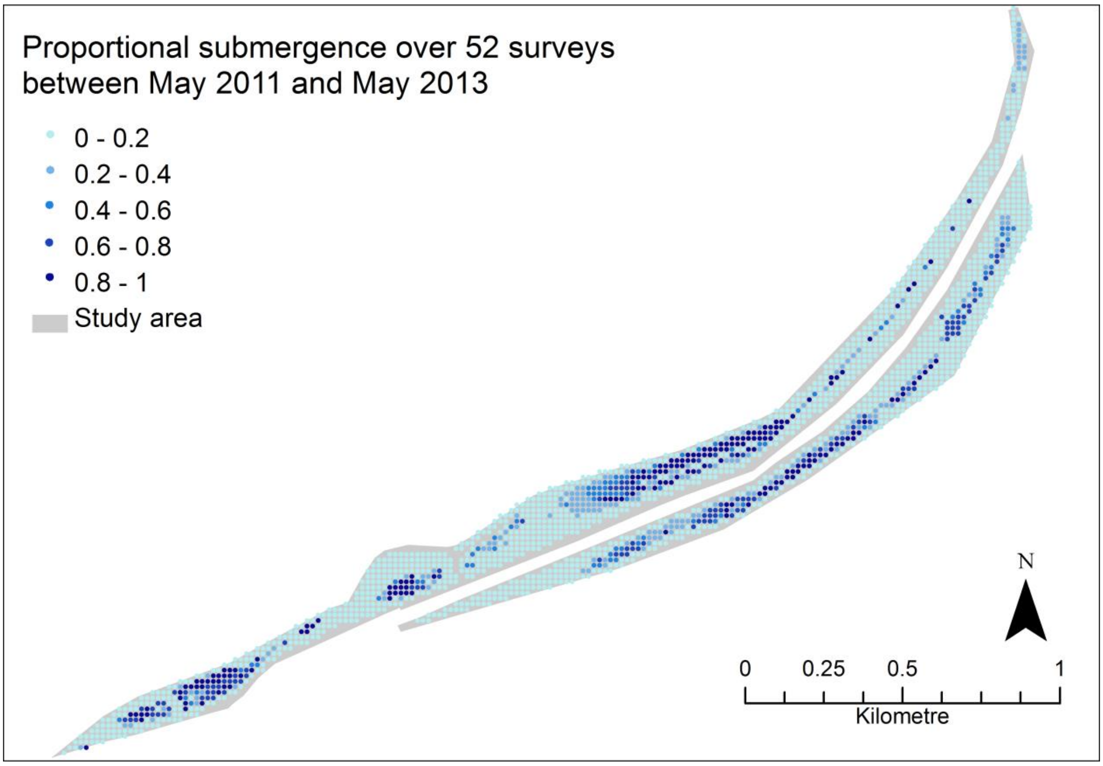

Figure 2.10. Proportional submergence at the Fishlake floodplain over the period 04/05/2011 to 19/04/2013. The area of standing water was sampled on 52 occasions at approximately two week intervals over this period.

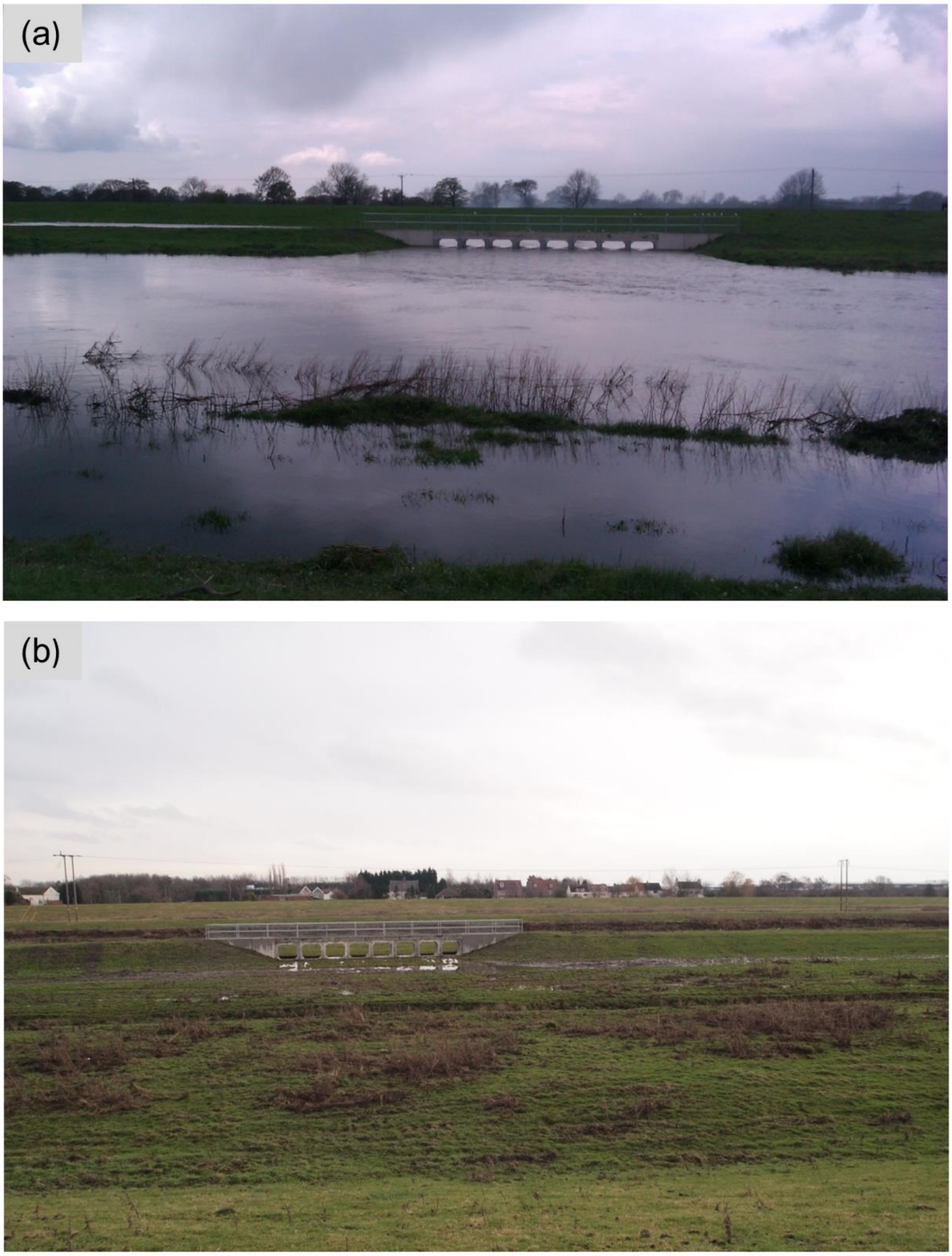

Figure 2.11. Comparion between high (a) and low (b) floodplain water levels at the eastern bank breach (see Figure 2.5). Photograph taken from the main flood defence bank, looking south. Photograph (a) was taken on the 20/04/2012, (b) was taken on the 10/01/2011.

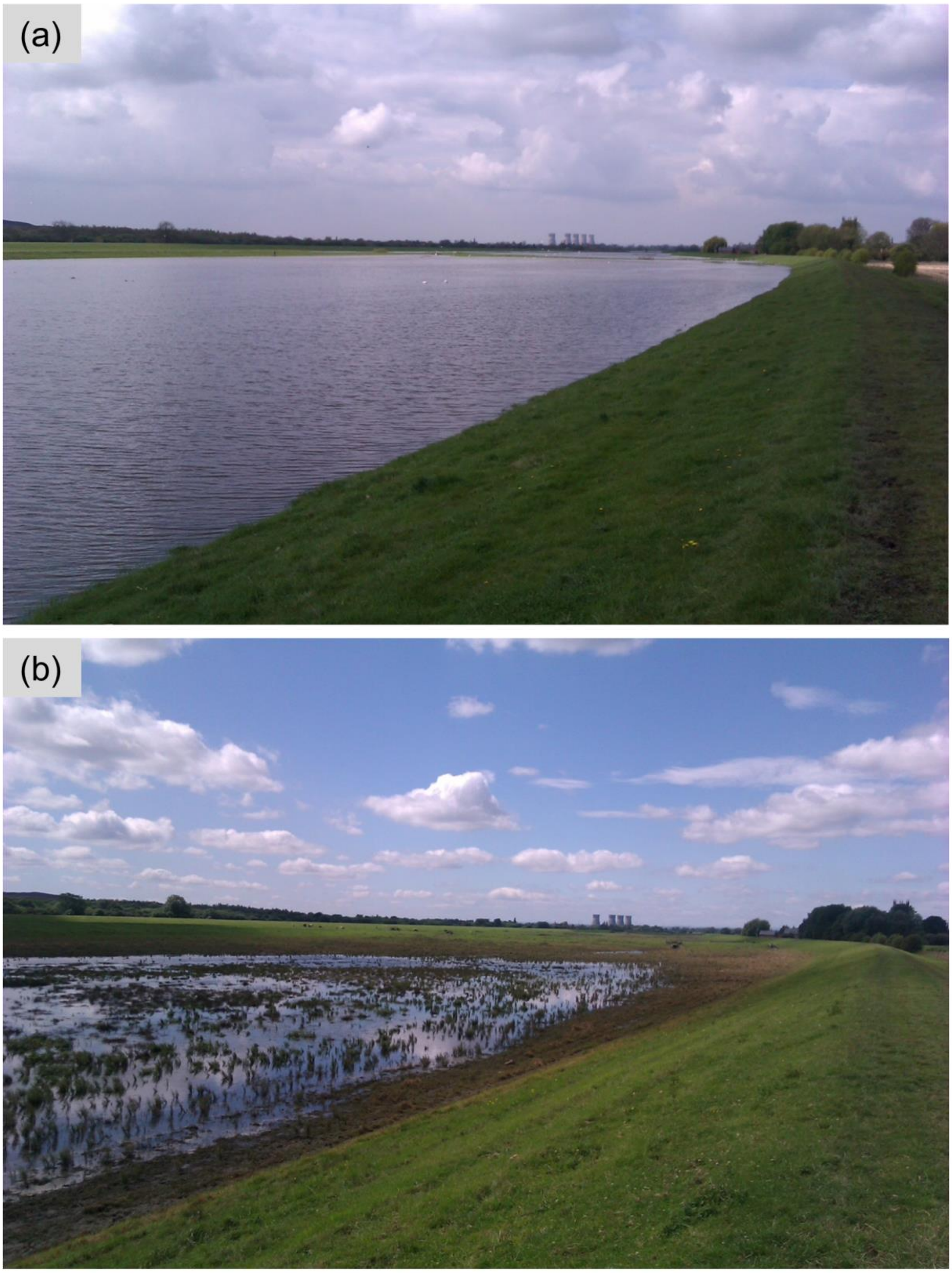

Figure 2.12. Comparison between high (a) and low (b) floodplain water levels in the northern bank of the floodplain. Photographs were taken from the point shown on Figure 2.2. looking south-west. Photograph (a) was taken on the 29/11/2012, (b) was taken on the 23/07/2012.

2.3. The research opportunity provided by the Fishlake floodplain restoration project

The Fishlake floodplain restoration project provides a good opportunity to study the practical application of ecosystem services to floodplain management because it is unusually well documented. The aims of the project were clearly defined in two design documents (Hiley et al. 2008; Environment Agency 2009), and there are five specific, measureable objectives that correspond to different ecosystem services. Many restoration projects do not carry out environmental or ecological monitoring (Bernhardt et al. 2007), and consequently it is rare to have baseline information such as a detailed LIDAR topography, or a time series of river level data that is specific to the study site. The Fishlake project also has baseline ecological datasets; water vole distribution data were collected in 2008 as part of the design evaluation process, while plant community composition was sampled by the University of Sheffield in 2009 as part of a monitoring agreement with the Environment Agency. These baseline ecological data allow the impacts of floodplain reconnection to be assessed directly in the following chapters. The baseline topographic data and continuous river level data were the core resources behind the hydrological monitoring and modelling described in section 2.2.3. The hydrological monitoring dataset and the associated hydrological model are invaluable throughout the following chapters of this thesis, as they allow ecosystem service responses to be analysed in relation to temporally and spatially detailed hydrological explanatory variables.

This chapter has drawn attention to a range of ecosystem services that are known to be provided at Fishlake, and there are further services that may be relevant in floodplain wetland systems. However, practical constraints mean that it is not feasible to analyse the full range of ecosystem services that a habitat can provide (Shaw 2012). The ecosystem services analysed in the following chapters were selected based on their relevance to the local area, and depending on whether baseline data was available. Ecosystem services were selected to ensure that a range of types of ecosystem services (provisioning, cultural, regulating) were represented. Chapters 3 and 4 of this study analyse indicators of biodiversity value, such as the presence of water voles and composition of the floodplain plant community. Biodiversity is of interest to the Environment Agency, who are a key stakeholder at Fishlake, and is given strong legal protection in the UK; water voles are a European Protected Species and floodplain grazing marsh is a Biodiversity Action Plan community type. Recreational experiences and indicators of recreational value are analysed in chapters 5 and 6, because recreation is a major use of European floodplains (Gren et al. 1995), and the Fishlake floodplain provides a recreational resource for the local population. An indicator of beef cattle production is analysed in chapter 6, as grazing is a major use of UK floodplains (Maltby et al. 2011), and a historical use of the Fishlake site. An indicator of flood storage volume is also analysed in chapter 6, as this is service has been prioritised at Fishlake in the past, and is of current interest in floodplain management (Baek et al. 2012).

3. Restoration of floodplain grazing marsh plant communities through increasing hydrological connectivity

This chapter investigates an ecological implication of restoring hydrological connectivity in floodplains; the short-term impacts on plant community composition. Plant communities are a vital component of habitats, and plants contribute a range of functions and can deliver many ecosystem services. This chapter focuses on the intrinsic biodiversity value of floodplain plant communities, and in particular floodplain grazing marsh community types. Floodplain grazing marshes are of high conservation interest in the United Kingdom, and extensive wetland areas like the Humberhead Levels have historically supported these communities. As discussed in Chapter 2, the improvement of floodplain grazing marsh was a specific target of the Fishlake restoration project.

This study made use of the baseline plant community dataset to analyse the responses of the community to restored hydrological connectivity at Fishlake. The chapter discusses the potential for similar hydrological restoration to enhance floodplain grazing marshes in other floodplains, and considers potential constraints on community development. The plant community dataset used in this chapter can be found in Appendix F.1 on the accompanying CD.

3.1. Abstract

Hydrological connectivity is a key feature of natural floodplains. Many floodplains in the United Kingdom have been disconnected from their adjacent rivers through the construction of flood defences and subsequently drained. Floodplain drainage has led to the loss of important habitats for conservation, such as floodplain grazing marshes in the United Kingdom. There is growing interest in restoring hydrological connectivity to floodplains, with the aim of restoring biodiversity. However, little is known about the impacts of floodplain reconnection on plant community composition, making the results of restoration projects unpredictable. Floodplain plant communities are strongly affected by hydrology, but there are a range of physical and ecological constraints that can prevent communities from developing along desirable trajectories. This study analysed changes in plant community composition at a case study floodplain restoration project, over a period of four years. Community composition decreased marginally in similarity to floodplain grazing marsh over the latter part of the study period. Changes in community composition were driven by the interaction between highly competitive species, which increased in abundance under relatively dry conditions, and moisture tolerant species, which increased in abundance under more frequent flood exposure. There was limited colonisation by new species, and this may have limited the trajectory of change towards floodplain grazing marsh. The results suggest that it is importance to restore functional connectivity that allows dispersal of floodplain species between habitat patches. Functional connectivity may not necessarily be achieved by the restoration of hydrological connectivity.

3.2. Introduction

Natural floodplains are species rich and ecologically diverse habitats, but up to 90% of the historical floodplain area in Europe has been converted to agricultural or urban use (Tockner and Stanford 2002). Many lowland river floodplains are no longer connected hydrologically to their adjacent floodplains, as embankments have been constructed to prevent flooding (Jungwirth et al. 2002). Surface connectivity between rivers and floodplains is important in maintaining mosaics of contrasting wetland habitats (Junk et al. 1989; Ward et al. 2002; Opperman et al. 2010), and river flooding can provide a dispersal pathway for organisms (Gurnell et al. 2008; Moggridge and Gurnell 2010). As a result of these hydrological mechanisms, natural floodplains typically support large numbers of specialist species (Tockner and Stanford 2002; Opperman et al. 2010). In the United Kingdom, floodplain grazing marshes, which include a range of semi-natural wet grassland and mire plant communities, are of particular conservation interest (Mountford 1994; Mountford et al. 2006). The restoration and creation of floodplain grazing marsh is now a conservation priority (JNCC 1995) and consequently there is increasing interest in reversing floodplain degradation through restoration projects that reconnect rivers and their floodplains hydrologically (Zsuffa and Bogardi1995; Toogood et al. 2008).

Floodplain restoration projects commonly target particular plant species and communities (Matthews and Spyreas 2010). To understand the likely impacts of floodplain reconnection, it is first important to know the hydrological requirements of the target community types (Härdtle et al. 2006; Toogood et al. 2008). The hydrological requirements of many floodplain grazing marsh communities have been described in detail (Wheeler et al. 2004), but communities commonly do not develop towards the desired targets following reconnection, as there a number of ecological mechanisms which can constrain or alter the trajectory of development (Trowbridge 2007; Matthews and Spyreas 2010). It is therefore important to understand not only the hydrological requirements of the target community, but how communities respond to changes in hydrological conditions, and how community development can be constrained.

Community composition changes as individual species change in abundance or colonise an area (Leeuw 1996). The local performance of a species under certain environmental conditions will depend on its environmental tolerances and ability to compete with neighbouring plants (Keddy 1992; Härdtle et al. 2006). These two mechanisms are closely related, because environmental conditions affect the balance of competition between species (Toogood et al. 2008). Species that are highly competitive under dry conditions (competitive specialists) may be negatively affected by flood events, for example if they cannot tolerate the resulting drought or root aeration stresses (Gowing and Spoor 1998), or are easily damaged by flood disturbance (Bornette and Amoros 1996). More frequent flooding may therefore benefit species that are more flood-tolerant, by decreasing competition from competitive specialists (Lenssen et al. 2004).

Hydrological connectivity can also provide opportunities for species to colonise a floodplain. Flood water can transport plant matter and seeds from habitats elsewhere in the river network (Hölzel and Otte 2001; Gerard et al. 2008), and these propagules can colonise habitat patches that have been cleared by disturbances (Gurnell et al. 2006). Long-distance dispersal of propagules can introduce species from connected wetland habitats (Malanson 1993), thus adding novel species to the local pool (Moggridge and Gurnell 2010).

Previous studies suggest that floodplain grasslands can respond to changes in hydrological regime over periods as short as a year (Toogood et al. 2008; Toogood and Joyce 2009), although even after decades they may not match the target communities (Donath et al. 2003; Bissels et al. 2004). Each of the above mechanisms may act to constrain a community from developing to a desired endpoint. The presence of highly competitive, flood-tolerant species such as woody invasives may prevent desirable species from establishing, even if hydrological conditions are suitable (Ho and Richardson 2013). Alternatively, there may be environmental factors other than hydrology that impact the performance of species, such as grazing pressure (Schaich et al. 2010b) or nutrient levels (Donath et al. 2003). Finally, desirable species may not be able to colonise the site, even if environmental conditions are suitable for them; for example if there are few reservoir populations that are able to disperse to the floodplain (Bischoff 2002). Hydrological conditions in floodplains are spatially and temporally variable (Ward et al. 1999; Tockner et al. 2000; Tockner and Stanford 2002; Tockner et al. 2010), so the relative impacts of these three mechanisms may be different depending on the hydrological conditions. A better understanding of the relative importance of these mechanisms, and their relationships with hydrology, may inform future floodplain reconnection projects.

Insights into the mechanisms that drive changes in plant communities can be gained by analysing the performance of species in relation to their characteristics, or functional traits (van Bodegom et al. 2006; Hedberg et al. 2013). Functional traits that describe the life history strategies of many UK species have been defined along the 3-way C-S-R spectrum (competitive ability, stress-tolerance, and ruderal ability) (Grime et al. 1995). Species tolerances to specific environmental factors such as soil moisture and soil nitrogen concentrations (Ellenberg 1988; Hill et al. 1999), and the seed dispersal mechanisms used by species have also been described (Grime et al. 1995). By analysing the performance of species in relation to these functional traits, the mechanisms that drive trajectories of community change can be inferred (van Bodegom et al. 2006; Gurnell et al. 2008).

It is logistically complex to analyse the responses of communities to changes in environmental conditions, because the financial costs of manipulating hydrological conditions can be high (Bernhardt et al. 2005) and community development may take long periods of time (Donath et al. 2003). Restoration projects provide opportunities to manipulate environmental conditions with some precision over large spatial scales (Walters and Holling 1990; Hammersmark et al. 2010). Analysis of current restoration projects could therefore help the mechanisms of change and constraints to development to be better understood, which may in turn inform future restoration attempts (Walters and Holling 1990; Bayley 1995; Michener 1997). However, there are relatively few studies of restoration projects despite the opportunities for research that they provide, because few restoration projects are monitored, either before or after restoration (Zedler 2000; Bernhardt et al. 2007). In this chapter the Fishlake floodplain restoration project is used as a model system to analyse changes in plant community composition following hydrological reconnection. The Fishlake floodplain provides a valuable opportunity for this research because a baseline record of the plant communities is available. Furthermore, a spatially-explicit model of hydrology at the floodplain allows changes in community composition to be related directly to hydrological conditions. This is relatively unusual for a wet grassland study; some previous floodplain studies have used proxies such as time since restoration (Vercoutere et al. 2007; Toogood and Joyce 2009) or topography (Trowbridge 2007) as explanatory variables.

Changes in the floodplain grassland plant communities were analysed over four years, separated into two periods (2009-2011 and 2011-2013). The similarity of each sampled community to the target of floodplain grazing marsh was analysed using a novel technique based on simulating floodplain grazing marsh communities. The functional trait responses underlying changes in community composition were then analysed by modelling the performance of species over the two pairs of years against six functional traits; competitor value and ruderal value (Grime et al. 1995), soil moisture and nitrogen tolerances (Ellenberg 1988; Hill et al. 1999), dispersal weight, and whether the species is known to spread vegetatively (Grime et al. 1995). Proportional flood exposure was also used as an explanatory variable, and interactions between flood exposure and all functional traits were analysed to investigate whether mechanisms acted differently under different hydrological conditions.

3.3. Methods

3.3.1. Vegetation and hydrological data

Plant community composition was analysed between 2009 and 2013 in the Fishlake floodplain, which is located on the River Don, near Doncaster in the United Kingdom (see Chapter 2 for an overview of the site). Engineering to create two bank breaches on the northern bank at Fishlake was conducted between July and August of 2009, and ditch dredging and reprofiling works were conducted on both banks of the river over the same period. A further culvert was added to connect the southern bank of the floodplain to the river in September 2011. These engineering works increased the degree of surface hydrological connectivity between the river and floodplain.

The plant community was first sampled at 28 plots of 400 m^2 each prior to any engineering in late June 2009. The sampling plots were chosen to represent the range of mainly terrestrial habitats (therefore excluding open water) that were present at the site at that time (Figure 3.1). Plots were clustered into transects for ease of relocation, but transect grouping is not considered to have any ecological relevance because the distance between transects is relatively low compared to the potential dispersal distance of plant propagules (Engström et al. 2009). Furthermore, the sites within each transect varied greatly in their topographic height, and thus their connectivity to the river. As a result, variation in hydrological conditions within each transect was likely to be greater than variability between transects. Sampling plot locations were recorded using a handheld Garmin E-trex H GPS and a combination of handheld maps, written notes, and photographs. The sample plots were re-surveyed during late June, in 2011, 2012, and 2013. In all surveys, all higher plant taxa were identified and assigned an abundance score on the ordinal DAFOR scale (Brodie 1985). Taxa were identified to species level except in the case of *Agrostis* and *Poa* grasses. Filamentous algae were also recorded. To ensure consistency in sampling plot relocation and taxonomy between sampling dates, at least one surveyor was conserved between consecutive samples. Although the floodplain plant communities were sampled on four occasions, most of the analyses in this study use the data from only three surveys and consider changes in community composition over two periods; between 2009 and 2011, and 2011 and 2013. The use of equal time intervals allows comparable changes in community composition to be analysed. The two periods are also functionally different, because part of the restoration work (on the south bank) was finished after the survey data were collected in July 2011. Additionally, these two periods were characterised by different rainfall and river level conditions; between the 2009 and 2011 surveys the river level was relatively low (there were no flood peaks above 5.3 m), while there were several larger flood peaks (six greater than 5.3 m) throughout the summer and autumn of 2012, and into the winter of 2012-2013 (Chapter 2; Figure 2.8).

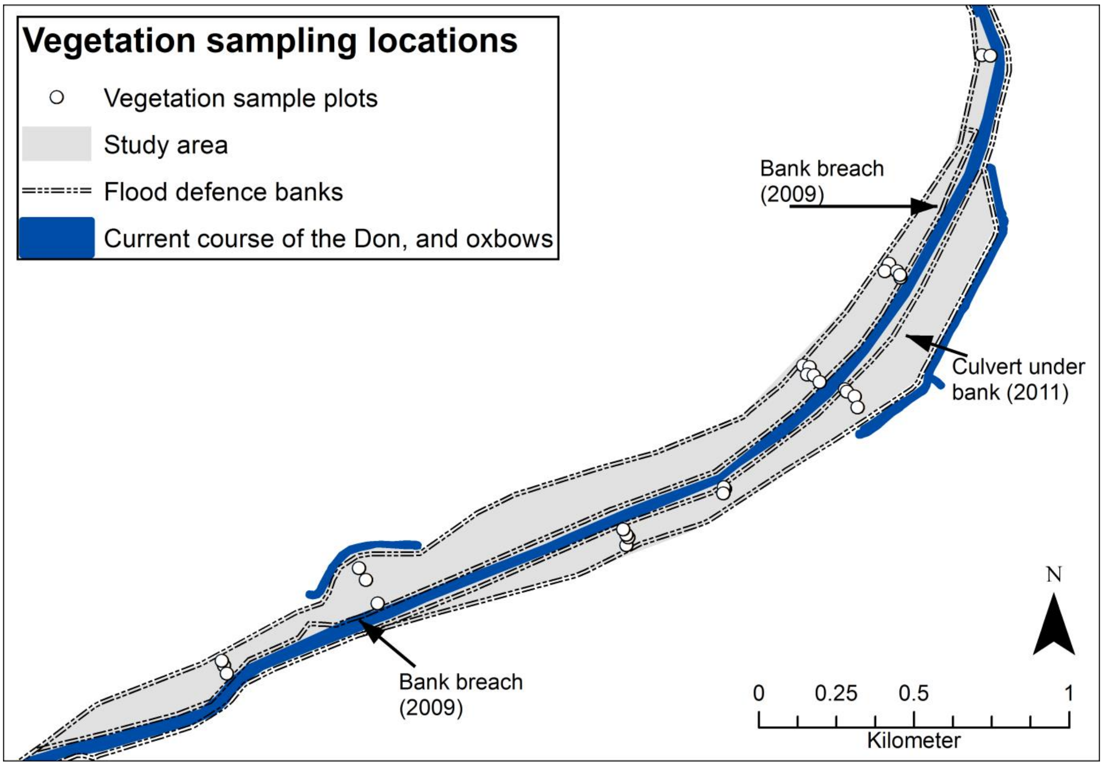

Figure 3.1. The study floodplain and River Don, showing locations of vegetation sampling quadrats.

A statistical model of flooding, similar to that described in Chapter 2, was constructed. The submergence of each survey plot was modelled separately using a binomial generalised linear model. The submergence of each plot was monitored at approximately two week intervals for two years, between 04/05/2011 and 19/04/2013 (Richards et al. in press). The proportion of each sampling plot that was underwater on a given hydrological monitoring date (n = 52) was modelled against the proportion of the preceding two weeks that the river water level was higher than the lowest point of the flood banks (see Chapter 2). The river level was monitored at an Environment Agency gauging station located at the site (see Chapter 2; Figure 2.2). Six of the resulting models were not statistically significant at alpha = 0.05, but were nonetheless used for subsequent modelling because they were likely to more realistically represent the relationship between flooding and wetness than the intercept-only model. Most of the insignificant models were either for very dry or very wet quadrats (Appendix B.1).

The regression models of flooding allowed the flood exposure of each sampling plot to be estimated from a time series of river water level data, for the periods of June 2009 to June 2011, and June 2011 to June 2013. From these time series an index of flood exposure was calculated for each sample plot; the proportion of the sample plot that was submerged for at least 50% of the time period. 50% was chosen as the threshold based on information in Wheeler et al. (2004), which describes the hydrological requirements of three floodplain grazing marsh community types (National Vegetation Classification types MG 4, MG 8, and MG 13). Among these three community types, MG 13 was judged to be the most relevant to the conditions at Fishlake, as MG 13 communities are commonly found “within a managed washland used for flood storage” (Wheeler et al. 2004). MG 13 requires a cumulative duration of flooding of 201 days per year (55% of the time) (Wheeler et al. 2004). There are a range of other floodplain grazing marsh classes that could establish at the Fishlake floodplain, but hydrological requirement data were not available for the majority of these communities. A 50% flooding duration threshold may be higher than the requirement for some floodplain grazing marsh communities, but this threshold is a robust indicator of the overall wetness of the sample plot. Sample plots that have a proportionally larger area that is submerged for at least 50% of the time are also likely to have a proportionally larger area that is submerged for at least 10% of the time. The relatively coarse temporal resolution of the submergence data (monitored once every two weeks) mean that lower thresholds could be sensitive to chance flooding events, so might not represent the longer-term hydrological conditions. Statistical differences in proportional submergence between the two periods were analysed using a paired t-test.

3.3.2. Temporal changes in community composition

Changes in plant community composition were assessed by comparing the similarity of the communities between 2009-2011, and separately between 2011-2013. The pairwise Gower dissimilarity (as implemented in the daisy function of the vegan R package) between the sampled communities was used to analyse the magnitude of changes in composition, because it is an appropriate index for ordinal data (Oksanen et al. 2012). The dissimilarity of the communities was visualised using non-metric multidimensional scaling. Permutational multivariate analysis of variance (non-parametric multivariate analysis of variance - NPMANOVA) of the dissimilarity matrix was used to assess whether the communities were significantly different between 2009 and 2011, and between 2011 and 2013. This procedure was carried out using the adonis function in the vegan R package (Oksanen et al. 2012), with the distance matrix modelled as a function of year, stratified by sample plot. A global NPMANOVA was first carried out, followed by pairwise comparisons between each consecutive set of years, using a Bonferroni-adjusted significance level ($p < 0.016$).

Beta diversity was calculated each year across the 28 sample quadrats. Beta diversity was quantified as:

$\beta = S/\alpha - 1$

Where β was beta diversity, S was the total species richness, and α was the mean species richness at each site (Tuomisto 2010). The species that had not previously been recorded that colonised the survey quadrats in 2011 or 2013 were recorded (colonisations), as well as species that went extinct between any consecutive pair of surveys (extinctions). The broad habitat requirements of these species (either submerged aquatic, terrestrial, or emergent) were assessed using the field guides Rose and O'Reilly (2006) and Haslam et al. (1982).

Plant community composition was analysed as the similarity of the sampled communities to the primary target of restoration at the Fishlake project; floodplain grazing marsh (Hiley et al. 2008). Swamp and aquatic communities were secondary targets of the project, but the project design document predicted that floodplain grazing marsh communities would show the greatest change in spatial extent (Hiley et al. 2008). Additionally, the survey quadrats were better placed to monitor changes in grassland vegetation than swamp or aquatic communities, because it was not possible to access areas where engineering was being conducted (including a number of pools and ditches) when the 2009 survey was carried out (Shaw et al. 2009). Floodplain grazing marsh is a broad habitat type of which there are a number of desirable target communities (Mountford et al. 2006), so desirability was analysed as the maximum similarity to a pool of National Vegetation Classification (NVC) community types that have previously been defined as floodplain grazing marsh. Mountford et al. (2006) define 40 NVC sub-communities as floodplain grazing marsh, but three of these community types have not been recorded within 100 km of Fishlake, so were excluded from the analysis as they were assumed to be unlikely to establish.

To analyse the similarity of the sampled communities to the 37 target NVC communities, the data contained in the NVC (the species richness of each community, and the relative frequency of each species in each community) (Rodwell 1991; Rodwell 1992; Rodwell 1995) were used to simulate a series of floodplain grazing marsh communities (50 simulations for each NVC community type). Similarity was measured as the number of species that were present in both the simulated community and the Fishlake sample, with a correction for the number that would be expected to match by chance given the numbers of species involved. This was calculated according to the equation;

$$C = M-(S/T \times F)$$

where C was the corrected similarity M was the number of matched species, S was the number of species present in the generated community, T was the total number of species present in the dataset (i.e. 1432), and F was the number of species present in the Fishlake sample. Similarity to each NVC community was quantified as the mean of the 50 replicates, and the score for the NVC community type that the sample was most similar to was taken as the index of similarity to the target of floodplain grazing marsh.

Temporal changes in similarity to the target communities were analysed as a repeated measures ANOVA conducted as a linear mixed-effect model using the lme4 R package (Bates et al. 2013), with statistical significance analysed using the lmerTest R package (Kuznetsova et al. 2013). The year of the survey was used as an explanatory factor. Post-hoc differences between pairs of years were analysed using Tukey's Honestly Significant Difference test, as implemented in the multcomp package for R (Hothorn et al. 2008).

3.3.3. Drivers of changes in community composition

Changes in community composition were investigated in more detail for the periods 2009 – 2011, and 2011 – 2013, to analyse the relative importance of flood exposure and species' functional traits in determining performance. Ordinal regression was used to model the number of DAFOR abundance classes that each species moved up or down between the two samples at each sampling plot (increases in class indicate increases in abundance, decreases in class indicate decreases in abundance). This performance index was modelled in response to six published functional traits; competitive specialisation, ruderal specialisation, and whether the species is able to spread vegetatively (from Grime et al. 1995), and moisture tolerance, nitrogen tolerance, seed weight (from Hill et al. 1999). The stress tolerance index from (Grime et al. 1995) was excluded because it is the inverse of the sum of the competitor and ruderal indices, so including it would create a singular model. The proportion of the plot area that would have been underwater for at least 50% of the intervening two years was also included as an explanatory variable (referred to as flood exposure). To analyse any relationship between functional traits and hydrological conditions, interactions between each functional trait and the flood exposure of the location were analysed. The performance of each species was modelled at each survey plot, and both species and survey plot were included as random effects (with random intercepts for both species and plot) to account for pseudoreplication. Ordinal regression models were constructed as cumulative link mixed models using the ordinal R package (Christensen 2013). In most statistical analyses in this study, an alpha level of 0.05 was applied when assessing significance. The only exception was the Bonferroni-adjusted NPMANOVA, as discussed above.

3.4. Results

There was a considerable range in proportional submergence between sample plots; most plots were relatively dry over both periods, but over 80% of the area of some plots was submerged for at least 50% of the time (Figure 3.2). There was a small but significant difference in hydrological conditions over the two periods; the proportion of the sample quadrats that was submerged at least 50% of the time was significantly greater between 2011 and 2013 than between 2009 and 2011 ($t = 2.73$, $df = 27$, $p = 0.011$).

A total of 81 plant taxa were recorded over the course of the surveys; these were mainly common semi-improved grassland species, but there were also a number of marsh and aquatic specialists. The dissimilarity of the sampled plant communities varied between years (Figure 3.3). The communities were significantly dissimilar between 2009 and 2011 ($F = 15.63$, $R^2 = 0.22$, $p < 0.001$), between 2011 and 2013 ($F = 2.1$, $R^2 = 0.037$, $p < 0.001$), and between 2009 and 2013 ($F = 2.18$, $R^2 = 0.03$, $p < 0.001$). Between 2009 and 2011 there were a larger number of plot-scale colonisations (299) and extinctions (215), than there were between 2011 and 2013 (112 colonisations and 189 extinctions). The species richness of the sampled communities differed significantly between years $F = 6.16$, $p < 0.001$). Species richness was significantly higher in 2011 than it was in 2009 ($z = 3.77$, $p < 0.001$), and was significantly lower in 2012 ($z = -3.19$, $p = 0.007$) and 2013 ($z = -3.46$, $p = 0.003$) that it was in 2011 (Figure 3.4A). The beta diversity of the communities at the sample plots was equal in 2009 and 2013, but dropped slightly in 2011 (Table 3.1)

The maximum similarity of the sampled plant communities to a desirable floodplain grazing marsh community was marginally significantly different between years ($F = 2.56$, $p = 0.06$). The only significant post-hoc difference between years was between 2011 and 2013, when the similarity to the target communities decreased ($z = -2.78$, $p = 0.03$; Figure 3.4B).

Whole-site species richness was highest in 2011, and lowest in 2009 (Table 3.1). The greatest number of new species were present in 2011, when 13 species were identified that were not present in 2009 (Table 3.1), and the greatest number of species went extinct from all of the survey quadrats between 2011 and 2013 (Table 3.1). Only two species colonised at more than one plot between 2009 and 2011 (*Callitriche stagnalis* and *Plantago lanceolata*, at two plots each) (Appendix B.2). In 2013 eight new species were present (Table 3.1), none of which were present at more than three plots. Seven of the 21 species which colonised the sample plots over either period were aquatic species, and all of these species colonised the sample plots between 2009 and 2011 (Appendix B.2). Eight species became extinct from all of the survey quadrats between 2009 and 2011, and 14 species became extinct between 2011 and 2013. Two of the species which went extinct were aquatic (Appendix B.3).

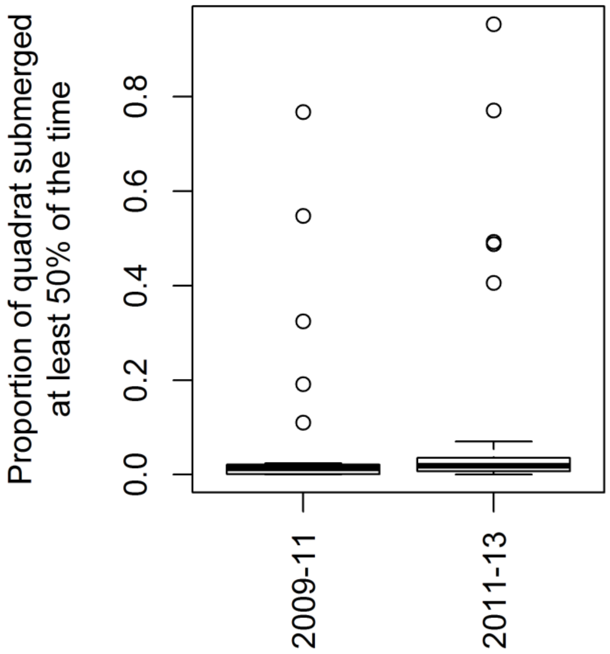

Figure 3.2. Proportion of the area of each study plot that was submerged for at least 50% of each two year period. Box and whisker plots show the median (bold line), interquartile range (box), and range within 1.5 × IQR (whiskers). Outliers are shown as circles.

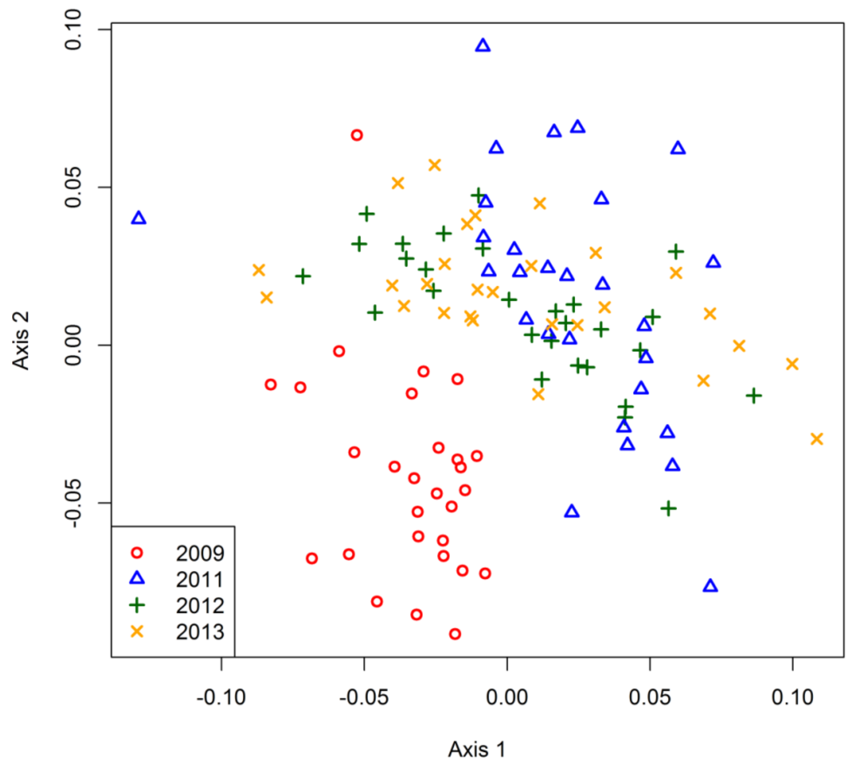

Figure 3.3. Non-metric multidimensional scaling of the plant communities sampled on four occasions. Different colours indicate the survey years.

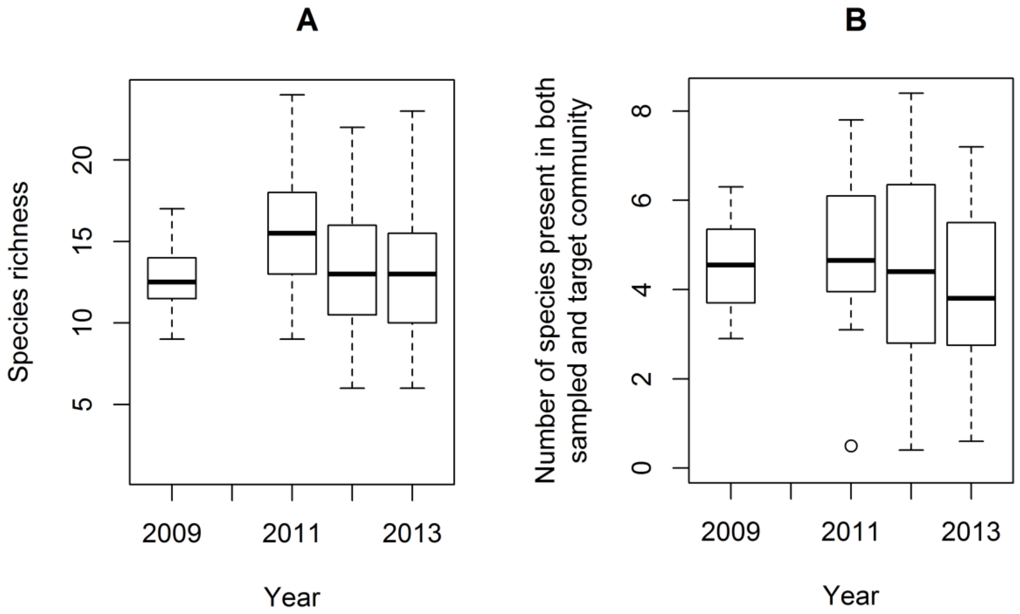

Figure 3.4. (A) Species richness at sampled communities over the years. Box and whisker plots show the median (bold line), interquartile range (box), and range within 1.5 × IQR (whiskers). Outliers are shown as circles. (B) Similarity of the 28 sampled plant communities to floodplain grazing marsh target community types. The score for each sampled community is the maximum number of species that were present in the sample that were also present in a floodplain grazing marsh target community.

Table 3.1. Species and community characteristics of the pooled survey quadrats over three study years.

Year	Total species richness	Beta diversity	Number of new species (compared to the 2009 survey)	Number of extinctions (compared to previous survey)
2009	59	3.6	NA	NA
2011	64	3.1	13	8
2013	60	3.6	10	14

Between 2009 and 2011, competitive specialists were significantly more likely to colonise or increase in abundance at study plots (Table 3.2), and moisture tolerant species were marginally significantly more likely to decrease in abundance (Table 3.2). All species were significantly more likely to decrease in abundance at plots with higher flood exposure (Table 3.2), and there was a significant positive interaction between flood exposure and moisture tolerance, indicating that moisture tolerant species were likely to perform better in plots with higher flood exposure (Table 3.2).

Between 2011 and 2013, moisture tolerant species were significantly more likely to increase in abundance (Table 3.3), and competitive specialists were significantly more likely to decrease in abundance (Table 3.3). All species were significantly more likely to decrease in abundance at plots with higher flood exposure (Table 3.3), and there was a significant positive interaction between flood exposure and competitive specialism, indicating that more competitive species performed better at plots with higher flood exposure (Table 3.3).

Table 3.2. Cumulative link mixed model of species performance over the period 2009 to 2011.

Variable	Estimate	Standard error	z	p
Competitive specialisation	3.03	1.45	2.09	0.03
Ruderal specialisation	1.97	1.29	1.53	0.12
Moisture tolerance	-0.25	0.14	-1.76	0.07
Flood exposure	-8.80	1.86	-4.71	<0.001
Moisture tolerance : Flood exposure (interaction)	1.29	0.27	4.63	<0.001

Table 3.3. Cumulative link mixed model of species performance over the period 2011 to 2013.

Variable	Estimate	Standard error	z	p
Competitive specialisation	-1.48	0.71	-2.08	0.03
Moisture tolerance	0.22	0.1	2.01	0.04
Flood exposure	-3.15	1.5	-2.09	0.03
Vegetative growth	0.58	0.35	1.64	0.1
Competitive specialisation : Flood exposure	3.03	1.54	1.97	0.04
Moisture tolerance : Flood exposure	0.34	0.21	1.58	0.1
Flood exposure : Vegetative growth (interaction)	-2.83	0.95	-2.95	0.003

3.5. Discussion

Restoration of hydrological connectivity between rivers and floodplains commonly does not result in widespread colonisation by novel species, at least over the first ca. 20 years (Bischoff 2002; Donath et al. 2003; Bissels et al. 2004; Rosenthal 2006; Gerard et al. 2008). Colonisation by new species can be limited if environmental conditions are not suitable for colonising species to establish (Eriksson et al. 1992). At the Fishlake floodplain it is possible that high nutrient levels, resulting from agricultural improvement (Donath et al. 2003) or grazing by cattle (Schaich et al. 2010b; Schaich et al. 2010c), limited the establishment of new species. However, many desirable floodplain grazing marsh species have relatively wide environmental tolerances (Mountford et al. 2006) and are commonly found in improved, grazed grasslands similar to the Fishlake floodplain (Crofts and Jefferson 1999; Hiley et al. 1998), so it seems more likely that colonisation is limited by low dispersal of new species. Colonising propagule densities may be low if there are few suitable source populations nearby (Bissels et al. 2004). The River Don and its associated catchment are highly modified (Shaw 2012), so patches of floodplain grazing marsh or other wetland habitats may be rare in the network (Buijse et al. 2002). Furthermore, any existing floodplain patches may be only rarely connected hydrologically to the river, thus reducing the chance of dispersal. It is notable that although only a few sample plots were colonised by new species, a third of the newly colonising species were aquatics. Aquatic taxa such as *Callitriche stagnalis*, *Elodea nuttallii*, and *Potamogeton* spp. are readily dispersed by river flow (Santamaría 2002), and can be found in open water habitats such as drainage ditches and ponds (Haslam et al. 1982). These habitat types are likely to be more prevalent in the heavily modified Don Catchment than high quality floodplain grazing marsh, which may explain the relatively high number of aquatic plants that colonised the sample plots.

The dispersal of plant propagules by flood events depends partly on the frequency and spatial characteristics of flood flow. In contrast to bankside riparian habitats, in which flow-mediated dispersal can play an important role in maintaining species distributions (Nilsson et al. 1991), floodplains are not continuously connected hydrologically to watercourses (Swenson et al. 2003; Reckendorfer et al. 2006), so opportunities for dispersal may be less constant. The deposition of propagules by flooding can also be spatially patchy, as the greatest numbers are typically deposited at strandlines (Vogt et al. 2004) where flood debris accumulates. Strandlines, particularly those left after larger flood events, are likely to be located at topographic high points. Higher areas are exposed to less frequent flooding, so may not be the most suitable areas of habitat for propagules of wet grassland or marsh plants. Patchily distributed propagules may eventually be dispersed more evenly over a floodplain area, but the delay and resulting desiccation may increase propagule mortality (Merritt and Wohl 2006).

This study showed significant changes in community composition over both study periods, despite the limited degree of colonisation by new species. This is consistent with previous findings that floodplain plant communities can respond dynamically to changes in environmental conditions over short periods (Toogood et al. 2008; Toogood and Joyce 2009). Plant species increased in abundance and colonised sample plots locally; either because persistent seedbanks were present (Geertsema et al. 2002; Gurnell et al. 2006; Rosenthal 2006), because of short-distance propagule dispersal by flood water, wind, or animals (Merritt and Wohl 2006; Rosenthal 2006), or because individual plants were able to spread vegetatively (McDonald 2001).

The performance of plant species depended on the hydrological conditions at each sample plot, because the functional traits of some species conferred greater competitiveness under dry conditions, while the traits of other species increased moisture tolerance (Lenssen et al. 2004). Species that performed well over the drier first period of the study had a high score for competitive specialisation, allowing them to out-compete species that invested in strategies such moisture tolerance. Conversely, moisture tolerant species decreased in abundance, probably because under dry conditions their adaptations for moisture tolerance did not confer benefits that outweighed the additional costs (Goldberg 1996). However, moisture tolerant species performed well at the plots with the largest submerged areas because of their competitive advantage under wetter conditions (Goldberg 1996; Lenssen et al. 2004). Almost the reverse pattern was observed between 2011 and 2013, as the species which performed well over this period were more moisture tolerant, and those which performed poorly were competitive specialists (except at the wetter sites). It is likely that the higher flood exposure between 2011 and 2013 caused the reversal in competitive performance compared to the period 2009 - 2011 (Goldberg 1996; Lenssen et al. 2004). The increase in the proportion of the sample plot that was submerged for at least 50% of the time was relatively small, but this index of flood exposure may disguise more subtle effects. It is likely that it is the shorter-term impacts of increased flooding which are likely to correlate with proportional submergence, such as the frequency of disturbance events or the duration of soil waterlogging, affected plant community composition.

It is interesting that the beta diversity between the plots did not increase over the survey period, because flooding is expected to create habitat heterogeneity, and thus variability in community composition (Ward et al. 1999). While the beta diversity remained the same, there was a greater interquartile range in both species richness and similarity to floodplain grazing marsh in 2013, compared to 2009 (Figure 3.4). This suggests that the communities developed along different trajectories depending on the hydrological conditions; some became more similar to floodplain grazing marsh, while others became very dissimilar. This is unsurprising given the selection for different functional traits in wet and dry sample plots.

Changes in plant community composition are constrained by the available species pool (Zobel et al. 1998), so if target species are not able to disperse to a floodplain then community composition can only change within certain limits. In the Fishlake floodplain it is likely that decreases in species richness, and particularly the loss of some competitive specialist floodplain grazing marsh species, caused the decline in similarity to the target of floodplain grazing marsh observed between 2011 and 2013. There are probably desirable floodplain grazing marsh species that could have tolerated the wetter conditions during 2011 – 2013, as floodplain grazing marsh is a broad category that includes both mires and drier, mixed grassland communities (Mountford et al. 2006). However, large numbers of moisture-tolerant floodplain grazing marsh species did not colonise the sample plots. It is therefore likely that the low colonisation rate by novel, desirable species limited the establishment of floodplain grazing marsh (McDonald 2001; Bischoff 2002; Bissels et al. 2004). If colonisation by desirable floodplain grazing marsh species is not facilitated at Fishlake in the future, and relatively wet conditions persist then it is likely that the similarity of the plant communities to the target will remain at the same level, or could even drop further. Alternatively, if drier years intersperse periods of more frequent flooding then similarity to floodplain grazing marsh may return to the level reached in 2011.

3.6. Conclusions

The restoration of hydrological connectivity in floodplains can have a substantial effect on plant community composition, and composition can change dynamically over short periods, depending on hydrological conditions. The rapid changes in community composition observed in this study suggests that floodplain plant communities could develop relatively quickly in response to hydrological conditions, but the trajectory of change was limited by the lack of floodplain grazing marsh species. This study draws attention to the importance of restoring not just the physical, structural connectivity between rivers and floodplains, but also the functional connectivity between populations of organisms (With et al. 1997; Wainwright et al. 2011). To understand functional connectivity it is important to look beyond a specific study site and consider the wider distribution of species in the landscape, and the dispersal mechanisms of particular species (Tischendorf and Fahrig 2000; Bélisle 2005). Future restoration attempts should not assume that target species will be able to disperse to a reconnected floodplain, but should consider the position of the site in the river catchment, and the connectivity to suitable source populations.

4. European water voles in a reconnected lowland river floodplain: habitat preferences and distribution patterns following the restoration of flooding

This chapter investigates an ecological implication of restoring hydrological connectivity in floodplains, specifically, the impacts on the presence and distribution of European water voles (*Arvicola terrestris*). Water voles provide biodiversity and cultural ecosystem services, as they are of high conservation interest in the United Kingdom and are popular with the general public. The Fishlake restoration project aimed to improve existing habitats and provide additional areas suitable for water voles, and ensure that the restoration had no negative impact on the population that was resident prior to restoration (Chapter 2).

This study made use of baseline data on the water vole population at Fishlake to assess changes in the distribution of the species following restored hydrological connectivity. To inform future management of floodplains for water voles, the habitat preferences of the species were characterised using six hydrological and vegetation characteristics. This study has been accepted for publication in Wetlands Ecology and Management, and is presented in this thesis as it will be published. The sections have been re-numbered and text has been formatted for consistency with the thesis, and all references can be found in the general references section. Water vole presence/ absence data and environmental explanatory variables can be found in Appendix F.2 on the accompanying CD.

4.1. Abstract

Water voles have suffered large population declines in the United Kingdom due to habitat degradation and predation by invasive American mink. Habitat restoration of floodplain wetlands could help to reverse this decline, but the detailed habitat preferences of water voles in these environments have not been well studied, and the impacts of restoration practices on water vole populations are not known. This study investigated the habitat preferences of water voles in a reconnected lowland river floodplain. The results show that water voles preferred wider water bodies, and taller and more diverse vegetation. The impact of flooding on water voles was also investigated by comparing their occurrence between two survey periods which were separated by large flood events, and by comparing distribution patterns before and after restoration. Contrary to previous reports, there was no observed negative impact of flood events on the water vole distribution, which has slightly expanded since the floodplain was reconnected to the river in 2009. Overall this study demonstrates that restored wetlands can provide suitable habitat for water voles, and provides guidance on some of the factors which should be considered when designing floodplains for water vole conservation.

4.2. Introduction

The European water vole *(Arvicola amphibius*) is a species of high conservation interest in the United Kingdom (UK), as it has suffered a sustained population decline since prehistory (JNCC 2008). The population is estimated to have declined by 99.9% since the Bronze Age (Jefferies 2003), and more recently, in a national survey of the UK, almost 70% of sites which were occupied when surveyed during 1989 - 1990 were unoccupied when resurveyed in 1996 - 1998 (Jefferies 2003). This decline has been due to the loss of suitable habitat, exacerbated in recent years by an increase in predation pressure from the invasive American mink (*Mustela vision*) (Woodroffe et al. 1990; Barreto et al. 1998a; Macdonald and Strachan 1999). Water voles occur in most freshwater habitats (Strachan and Jefferies 1993), but the loss of floodplain wetlands may have been particularly damaging to the species (Barreto et al. 1998a). Floodplains have undergone extensive declines due to river impoundment and land drainage; it is estimated that 90% of the major rivers in Europe no longer support natural floodplain wetlands (Malmqvist and Rundle 2002; Tockner and Stanford 2002).

The loss of floodplain wetlands may have had a greater impact on water voles than the loss of river and stream habitats because of the interacting effect of mink predation. It has been shown that healthy water vole populations can survive in wetlands despite the presence of mink, suggesting that these habitats may provide refuges from predation (Macdonald et al. 2002; Carter and Bright 2003). This may be because wetlands are difficult for mink to travel in, making foraging more costly than it is in other habitats (Macpherson and Bright 2010a). If wetlands enable water voles to survive predation then restoring these habitats is likely to be a more sustainable conservation strategy than attempting to control mink through culling (Barreto et al. 1998a; Barreto et al. 1998b).

Historical river impoundment and land drainage have isolated many floodplains from adjacent river channels, causing degradation of their associated wetlands (Jungwirth et al. 2002; Brandolin et al. 2012). Restoration practices commonly involve removing or lowering flood defences, thus reconnecting the floodplain with the river (Fischenich and Morrow 2000; Florsheim and Mount 2002; Breithaupt and Khangaonkar 2011). This restored flooding creates a mosaic of habitats (Ward et al. 1999; Buijse et al. 2002), including some that are known to be suitable for water voles; such as standing water, marsh and bankside terrestrial habitats (Strachan and Jefferies 1993). However, it is not clear whether all component floodplain habitat types are equally suitable for water voles. Whilst habitat preferences of water voles have been well studied in rivers and streams (Lawton and Woodroffe 1991; Barreto et al. 1998a; Aars et al. 2001; Telfer et al. 2001; Bonesi et al. 2002), wetland studies have been fewer, and have focused on habitat selection at the landscape scale (Strachan and Jefferies 1993; Macdonald et al. 2002), so only have analysed relatively broad habitat characteristics such as patch size and adjacent land use. Additionally, wetland studies have not investigated the impact of hydrological variability on habitat preferences. Hydrology is of particular importance in relation to restoration projects as the frequency and magnitude of flooding are altered by floodplain reconnection, and can be predicted and controlled by the design of the flood bank breaches (Fischenich and Morrow 2000; Hammersmark et al. 2005).

Previous studies of water vole habitat preferences in river and stream networks have considered associations with some hydrological variables. For example, measurements have been made of stream depth, width and flow velocity (Aars et al. 2001; Telfer et al. 2001; Bonesi et al. 2002; Chen 2010), and flow features, channel features, and substrate characteristics have been counted and classified (Barreto et al. 1998a). Whilst the findings of these studies have some application to floodplain wetlands, watercourse networks can be very different in structure compared to floodplain wetlands. Rivers and streams provide linear strips of habitat often bordered by steep banks, whereas wetlands commonly provide wider habitat patches made up of nonlinear water bodies that vary in their connectivity to the river and degree of water level variability (Tockner et al. 1999; Carter and Bright 2003). Moreover, the hydrological measurements made in previous studies represented single points in time, so did not account for the temporal variation that can be present, particularly in wetlands that flood periodically. Water level fluctuations are considered to have a negative impact on habitat suitability for water voles (Strachan et al. 2011), and there have been reports of large flood events causing mortality or displacement from burrows (Halliwell and Macdonald 1996; Macpherson et al. 2003; Moorhouse et al. 2009). However, whilst one study in a reedbed included some water level monitoring (Hardman and Harris 2010), none have investigated the impact of hydrological variability on habitat suitability for water voles.

Here, the habitat preferences of water voles within a restored floodplain are analysed using presence data recorded over two summer seasons. Probability of presence is modelled as a function of six environmental variables, including four hydrological factors drawn from a time series of water level measurements. This study system allows further investigation of the impacts of hydrological variability on water voles. Several flood events occurred between the two survey periods, allowing us to consider the impact of flooding on water vole site occupancy. Baseline data collected prior to the restoration project also provide a rare opportunity to consider the longer-term impact of floodplain reconnection on water voles. The objectives of this paper are to (1) analyse the habitat preferences of water voles in a floodplain wetland, and (2) discuss the longer-term implications of hydrological restoration on floodplain water voles.

4.3. Materials and methods

4.3.1. Study site

The Fishlake wetlands are a 0.8 km^2 area of lowland river floodplain located in South Yorkshire in the United Kingdom (Figure 4.1A). The site lies adjacent to the River Don and is part of the Humberhead Levels, a region that historically contained extensive floodplains (Van De Noort 2000), and has recently been awarded funding for landscape-scale conservation and ecological restoration as a part of a national scheme (Natural England 2013). The floodplain at Fishlake has been used as a flood storage area during extremely high flows since it was embanked and drained in 1943, and some standing water has always been present in the form of drainage ditches and ponds. The flood defence banks were lowered by approximately 1.5 m at two locations in 2009, and a culvert connecting the river to a floodplain pond was added in 2011. These bank breaches were put in place to increase the connectivity between the river and the floodplain, and overbank flooding now occurs at least thirty-six times a year, compared to between zero and twelve times a year previously (Hiley et al. 2008). The River Don in this area is tidal although not brackish, therefore flooding occurs at spring tides as well as during high river flows (Hiley et al. 2008). A mink scat was found adjacent to the site in September 2011, and other water vole predators such as red fox (*Vulpes vulpes*), common kestrel (*Falco tinnunculus*) and grey heron (*Ardea cinerea*) were observed at the site over the study period.

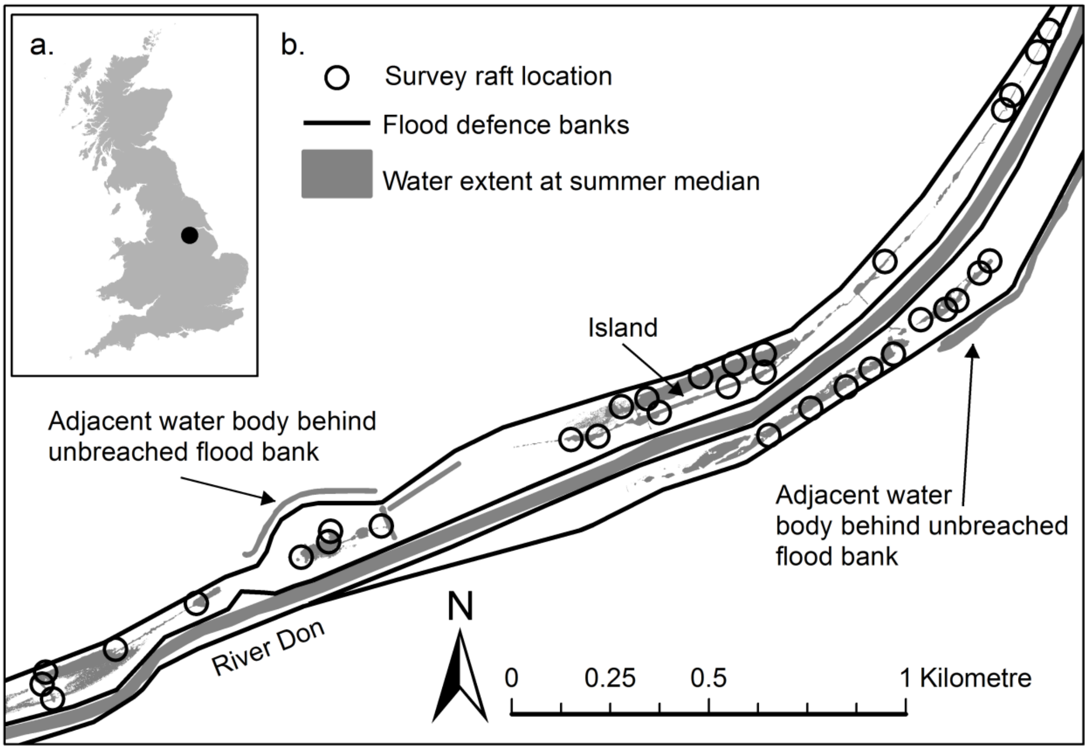

Figure 4.1. (a) (Inset) Location of the study in the UK marked by a filled black circle. (b) (Main) Locations of survey rafts within the Fishlake floodplain. Water extent including the River Don is indicated by grey shading. Flood defence banks are indicated by solid black lines. Raft locations are indicated by open black circles.

4.3.2. Water vole occurrence at artificial floating platforms

The presence or absence of water voles was established using 0.25 m^2 plywood and polystyrene survey rafts. These floating platforms provided feeding and latrine sites, which were used by water voles resident in the area (Hardman and Harris 2010). Thirty-four rafts were placed at randomly selected locations along all water bodies across the study site, at least two metres from the water's edge (Figure 4.1B). The river itself was not monitored during this survey due to limited access, and because it was not expected to provide suitable habitat; the banks are more than 20 m apart and the water level fluctuates by at least two metres twice daily (Bonesi et al. 2002; Strachan et al. 2011). Survey rafts were deployed for one week and then checked for field signs on four occasions at one week intervals between September and October in 2011. This procedure was repeated in 2012. Water vole droppings and feeding remains are highly distinctive; only juvenile field signs, which are rarely found in separate locations from adult signs, may be easily confused with those of other species (Strachan et al 2011). Any adult droppings and feeding remains present during a visit were recorded and then removed from the raft.

4.3.3. Habitat characterisation

Vegetation height was measured near each raft position, at six locations spaced at 1 m intervals along a transect running perpendicular to the bank. Height was measured by dropping a 30 cm diameter polystyrene disc onto the vegetation and measuring the distance between where the disc came to rest and the ground (Stammel et al. 2003). A 20 m radius around each survey raft was used to calculate additional habitat characteristics, as this corresponded to a 40 m stretch of a linear water course, approximating a water vole's daily home range in marsh habitat (Moorhouse and Macdonald 2005). Plant taxa present in the water and bankside area (on land within 3 m from the water's edge) were recorded in the vicinity of each raft in September 2012. Taxa were identified to species level except in the case of sward grasses (all grasses except reed sweet-grass *Glyceria maxima*, which is a large, reed-like species), and the relative abundance of each species was estimated using the DAFOR scale (Brodie 1985). Water voles are generalist herbivores so are likely to feed on, and use for cover, the majority of the species present at a site (Strachan and Jefferies 1993). Simpson's diversity index was calculated for the plant community sampled at each site (Krebs 1999), using the vegan package for R (Oksanen et al. 2012) in order to quantify the wider range of foraging opportunities and better cover that may be provided by more diverse vegetation (Lawton and Woodroffe 1991). Plant taxa received separate DAFOR scores where they were present both within water bodies and on land (i.e. they were recorded as separate taxa), in order to distinguish the structural evenness of habitats with aquatic, emergent and bankside vegetation.

The extent, depth and location of floodplain water bodies were mapped using a novel method developed for the Fishlake study site. Shoreline coordinates around each water body were recorded using a global positioning system (GPS) during a site walkover, approximately once every two weeks between 04/05/2011 and 16/10/2012. These points were then cross-referenced with a high-resolution digital elevation model (DEM) (scale: 1 pixel = 0.0625 m^2) derived from UK Environment Agency-supplied airborne light detection and ranging (LIDAR) data, and survey data collected using a Leica TCRP1205 robotic theodolite and electronic distance meter. This cross-referencing process generated a number of estimated heights of the water level on each occasion. The site was divided into topographically separate hydrologic units, and the water level within each section was defined as the mean of the water level height estimates that were recorded within it. All areas lower than these mean values were then classified as being underwater. This procedure provided a series of measurements of the depth and extent of standing water between 04/05/2011 and 16/10/2012.

The topographic and water level data were combined within a geographic information system (GIS). The area within a 20 m radius of each survey raft was again used to calculate the hydrological habitat characteristics. The maximum wetted width and water depth within the 1256 m^2 area surrounding each raft was calculated as the median of the water level measurements recorded over the two summer periods (May-October in each year, n = 24).

To provide an index of water level variability, the range in water depth at each raft site (maximum-minimum) was calculated across all of the temporal water level records (n=38). To assess the impact of water level variation at a finer temporal resolution, the largest increase in water depth between two consecutive measurements, calculated over the entire water level monitoring period (n=38), was used as an indicator of cumulative flood magnitude over each 2 week period.

4.3.4. Water vole distribution before and after restoration

Water voles were present in this study system before the restoration work was initiated. A survey carried out in 2008 (one year before restoration) by the local Wildlife Trust established presence using a field sign search within and around (within 3 m of) all water bodies, supplemented in some areas by the use of artificial survey rafts. Unfortunately, the locations of survey rafts yielding no field signs were not always recorded, but positive results can still be used to map the known distribution of water voles at the time of the survey.

A similar hand search for field signs was carried out in 2011, allowing some comparison of the water vole distribution across the whole floodplain before and after reconnection with the river. The effort invested in the 2008 and 2011 surveys was comparable; two experienced surveyors searched continuously in all bankside and aquatic habitats at a slow walking pace.

4.3.5. Data analysis

The proportion of survey visits to a raft when water vole field signs were found was used as the indicator of habitat preference. It was not always possible to survey every raft, as some came loose and were either lost or moved far from their original location. The activity of water voles at the raft locations was modelled as a binomial proportion in a generalised linear model (GLM), weighted to take account of differences in survey effort (the number of successful surveys), against habitat characteristics. Candidate predictor variables were compared pairwise using Pearson's correlation coefficient to quantify possible confoundment between habitat variables.

The maximal model including all six habitat characteristics was simplified through a backwards stepwise procedure using Akaike's Information Criterion, corrected for small sample sizes (AICc), to compare the models at each stage (Burnham and Anderson 2002). The proportion of the error variation explained by the model was assessed using the likelihood ratio R-squared (Menard 2000). Map processing was carried out using ArcMap 10 (ESRI 2011) and the raster package for R (Hijmans and van Etten 2012), and all statistical analyses were performed in R 2.15. (R Core Development Team 2012). An alpha value of 0.05 was used to evaluate the significance of statistical tests throughout this study.

4.4. Results

4.4.1. Habitat characterisation

The water bodies sampled were generally still-water, although some flowed periodically during flood events. They varied in configuration from extensive pools to linear drainage ditches, and from highly vegetated marshes dominated by emergent *G. maxima* to more open water bodies dominated by aquatic vegetation. Water body width was significantly positively correlated with water depth (Pearson's $r = 0.45$, $t = 2.84$, $df = 32$, $p = 0.007$), range in water depth (Pearson's $r = 0.42$, $t = 4.99$, $df = 32$, $p < 0.001$), and largest flood magnitude (Pearson's $r = 0.64$, $t = 4.72$, $df = 32$, $p < 0.001$). Range in water depth was also significantly positively correlated with largest flood magnitude (Pearson's $r = 0.97$, $t = 22.53$, $df = 32$, $p < 0.001$). These correlations were not unexpected, but as these variables have different potential ecological impacts on water voles, all were included in the maximal model.

Water levels within the floodplain varied over time. The summer of 2011 was characterised by low water levels, whilst a slightly higher level was maintained throughout the rest of the period (Figure 4.2). The water level on the site was slightly higher in the 2012 survey period (the mean level on the north bank was 2.37 m and on the south was 2.41 m) than during the 2011 surveys (the mean level on the north bank was 2.04 m, and on the south was at 2.24 m), and in both years some flooding occurred between water vole surveys, therefore raising the water level. Several large floods occurred in May and July 2012 on both sides of the river, although water levels on the northern bank tended to be higher due to the design and position of the flood bank breaches, and the efficiency of drainage (Figure 4.2). Additional large floods occurred in the winter of 2012, after the end of the water vole survey period. For reference, a water level of 4 m is enough to flood the entire site, leaving continuous stretches of water and only a few islands of terrestrial habitat within the flood banks shown in Figure 4.1.

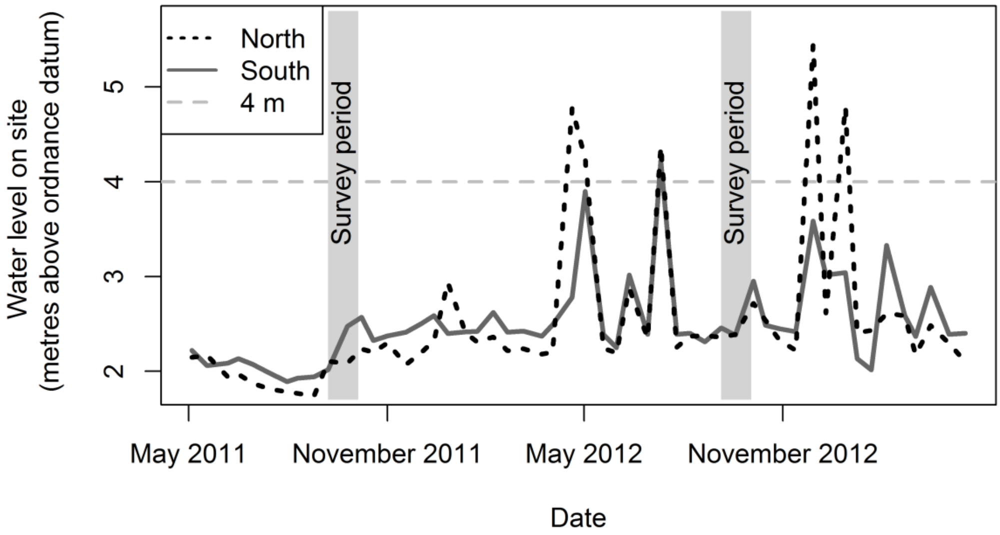

Figure 4.2. Water level in floodplain wetlands on the northern (black dashed line) and southern (grey solid line) banks of the River Don, measured at two week intervals over 18 months. At a water level of 4 m (light grey dashed line) most of the study area within the flood defence banks is underwater. Water vole survey periods are indicated by grey shaded areas.

4.4.2. Habitat preferences

Water voles were found on both sides of the river, at 8 sites in 2011, and at 8 sites in 2012 (Figure 4.3). There was consistency in site occupancy between the 2 years, with 6 sites occupied in both 2011 and 2012. It is not surprising that there was some change in site occupancy between years, as water vole populations can show high rates of turnover (Aars et al. 2001; Telfer et al. 2001). Therefore the results of the 2 years were pooled for model construction, with proportional activity calculated over all successful survey visits (maximum number of successful visits = 8). Analysis of the two years separately produced comparable results to the pooled analysis.

The maximal model fitted to these data was simplified to one comprising four parameters; mean vegetation height, plant diversity, water body width, and flood magnitude (Table 4.1). The likelihood ratio R-squared for the final model was 0.48. The probability of water voles being present increased significantly with vegetation height, plant diversity, and water body width (Table 4.2). The probability of presence of water voles decreased with increasing flood magnitude, but the effect of this factor was only marginally significant (Table 4.2).

4.4.3. Temporal changes in water vole distribution

The number of sites occupied did not change between the two years (Figure 4.3), despite the flood events that submerged the study area during 2012 (Figure 4.2). Additionally, the field signs mapped during hand searches showed little change in distribution between 2008 and 2011, although there was a possible, albeit subtle sign of expansion into a new area of the floodplain (Figure 4.4). There may also have been subtle range contraction in the drainage ditches that are adjacent to the restored floodplain area (Figure 4.4). Given the known stochasticity of water vole site occupancy, and the precision of the survey method, however, it is not possible to attribute this slight difference between years to habitat restoration.

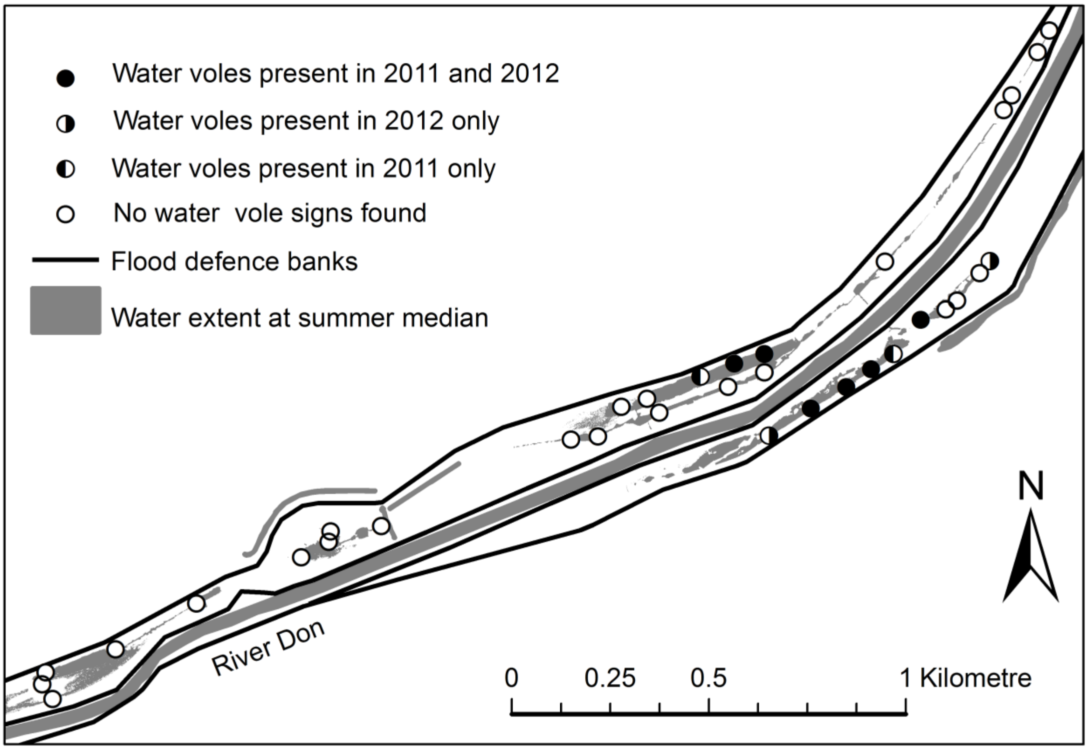

Figure 4.3. Presence/ absence of water voles at survey rafts within the Fishlake floodplain over two years. Water extent including the River Don is indicated by grey shading. Flood defence banks are indicated by solid black lines. Presence of water voles at each raft location is indicated by the filling of half of the open circle. A filled left hand side of the open circle indicates presence in 2011, a filled right hand side indicates presence in 2012.

4.5. Discussion

The water voles in the study floodplain showed preferences for wider water bodies, taller vegetation, and greater plant diversity. These preferences are similar to those observed in rivers, streams, and drainage ditches (Lawton and Woodroffe 1991; Barreto et al. 1998a; Aars et al. 2001; Telfer et al. 2001; Bonesi et al. 2002; Chen 2010), and the presence of water voles in this floodplain is consistent with the hypothesis that wetlands can provide the species with good quality habitat (Barreto et al. 1998b; Carter and Bright, 2003). Previous studies have focused on the value of wetland habitats such as reedbeds in providing cover from predators, specifically mink (Carter and Bright 2003; Macpherson and Bright 2010a). However, water vole habitat preferences are driven by their basic requirements for shelter and forage, and their sensitivity to disturbance from larger animals, as well as their sensitivity to predation (Strachan and Jefferies 1993; Strachan et al. 2011). It is therefore likely that the habitat characteristics selected by water voles in this floodplain wetland provide them with a combination of benefits.

Water voles are generalist herbivores that require large amounts of vegetation (Strachan and Jefferies, 1993; Moorhouse et al. 2008). A greater density of plant biomass, as indicated by taller vegetation (Redjadj et al. 2012), should allow more efficient resource collection because the time spent travelling between forage patches should be reduced. Water voles forage in emergent wetland vegetation, or on land close to the water's edge (Lawton and Woodroffe 1991; Macdonald and Strachan 1999; Moorhouse et al 2008). Wider water bodies can therefore provide a greater area of suitable habitat within the same distance from the burrow, provided that there is sufficient cover and forage present in the centre of the water body. This should allow water voles living in wider water bodies to forage more efficiently by reducing their travel time. However, water voles do not regularly travel large distances in search of food (Sah 1998; Moorhouse and Macdonald 2005) and the resource environment for grazing herbivores is expected to be continuous rather than patchy (Senft et al. 1987), so the gains in foraging efficiency alone may be marginal. Greater plant diversity may provide foraging benefits for water voles because a varied diet can better provide the required nutrients and encourage an increased rate of grazing (Provenza et al. 2009).

The habitat preferences of water voles can also be affected by interactions with other species such as cattle. Areas with tall vegetation are likely to have experienced less cattle grazing, which is likely to be beneficial for water voles because heavy trampling can destroy burrows (Macdonald and Strachan 1999), and physical disturbance by grazers can interfere with foraging (Barnard 1980). Avoidance of cattle may also be easier in wider water bodies, as cattle avoid aquatic habitats (Ballard and Krueger 2005) and do not venture far from the bank when drinking (Daniel Richards, personal observation).

Water voles have many predators (Forman 2005), and it has been proposed that wetland habitats may offer some protection from predation (Carter and Bright 2003; Macpherson and Bright 2010a). Tall vegetation may provide cover from raptors because it presents a visual barrier when viewed from above (Korpimäki 1985; Aschwanden et al. 2005). Preferences for diverse vegetation could also be driven by the risk of aerial predation, as variation in plant form and leaf type could be expected to provide better protection. Such structural diversity in vegetation was not measured directly in this study, but has previously been shown to be preferred by water voles (Lawton and Woodroffe 1991). The combination of tall and diverse wetland vegetation can also present a locomotive barrier to mustelids such as mink (Macpherson and Bright 2010a). This barrier may deter mink from foraging in tall, diverse vegetation if adjacent habitats provide more suitable opportunities, and similar barriers have been shown to slow down predators when they are in pursuit (Wywialowski 1987). Wider water bodies may also provide greater protection from predators; it is known that mink forage more frequently at the edges of reedbeds than they do in the centre (Macpherson and Bright 2010b), and large expanses of water can deter terrestrial predators such as red fox (Lokemoen and Woodward 1992). As discussed previously, wider water bodies provide the required area of suitable foraging habitat within a shorter distance from the burrow, thus reducing the distance to shelter.

Wider water bodies in this sample tended to be deeper, which may be beneficial for water voles because diving is a common predator evasion technique (Woodroffe et al. 1990; Macpherson et al. 2003). It is likely that wide, deep water bodies are more permanent, and therefore provide habitat more consistently throughout the year. This would mean that resident water voles are less likely to be forced to disperse due to lack of water. Another component of hydrological variability is the magnitude of increases in water level. Large increases in water level are expected to have a negative impact on habitat suitability because these events can cause burrow networks to flood, which in turn can cause displacement of individuals into unfamiliar habitats, chilling, and an increased risk of predation (Strachan et al. 2011; Macdonald and Strachan 1999). In this study there was a suggestion that occurrence of water voles became less likely as flood magnitude increased. However, this effect was only marginally significant in the final model, possibly because flood magnitude was confounded by water body width. The variation in flood magnitude observed was substantial (Table 4.1) but relatively small when compared to variation between different watercourses (Poff et al. 1997; Sparks et al. 1998). It could be that at this scale of variation the benefits provided by wider water bodies are greater than the risks associated with increasing water level variability.

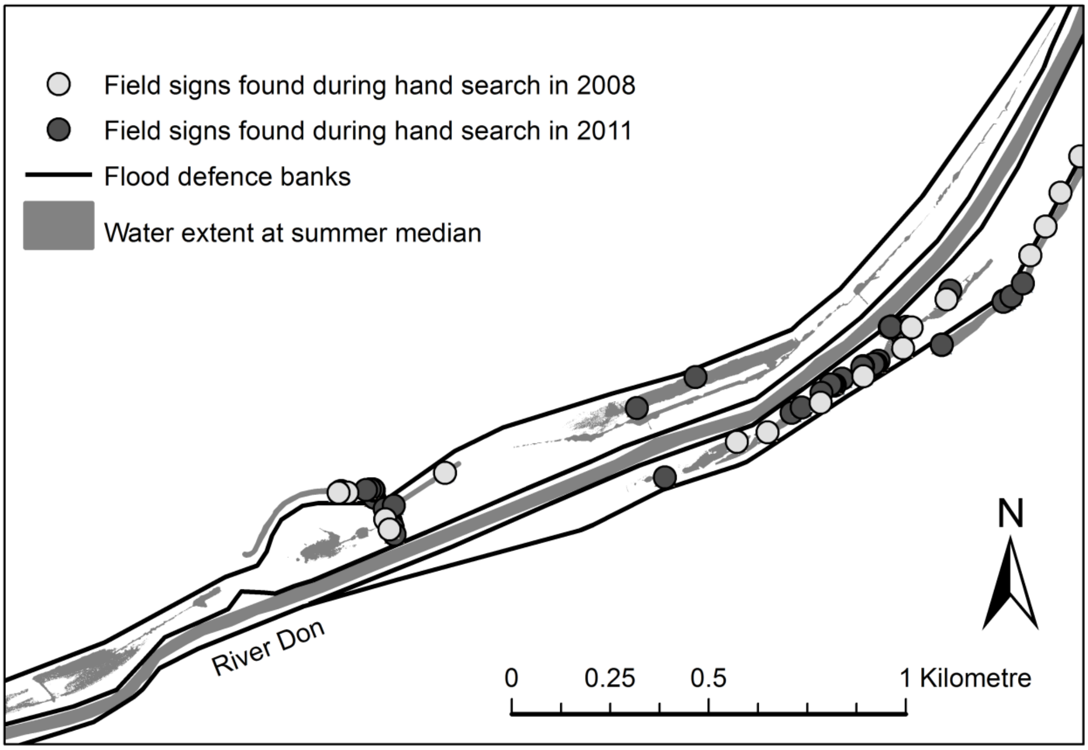

Figure 4.4. Locations of water vole field signs found during hand searches before and after floodplain reconnection. Water extent including the River Don is indicated by grey shading. Flood defence banks are indicated by solid black lines. Field signs found in 2008 are indicated by light grey filled circles, field signs found in 2011 are indicated by dark grey filled circles.

There was no evidence of any change in site occupancy or distribution between different years. The same number of sites was occupied in 2011 and 2012, despite flood events that left the floodplain submerged for several days. Additionally, there was no noticeable change in distribution pattern between the field signs recorded before and two years after restoration, despite the increase in flooding following reconnection. This information is limited but the pattern observed is in contrast with previously documented flood events. Macpherson et al. (2003) report that flooding of a similar scale, causing complete submersion of a reedbed wetland, "devastated" the local water vole population, leaving no sampling transects occupied. Additionally, a flood event in a lowland river system caused mortality of almost 90% and resulted in the failure of a reintroduction attempt (Moorhouse et al. 2009). A better understanding of the individual responses of water voles to flooding would help to assess why floods in some situations are more damaging than in others. However, there are some characteristics of the study site and hydrological regime that could explain this finding and inform the design of future floodplain reconnection projects. At Fishlake there are water bodies adjacent to the reconnected floodplain area (Figure 4.1) which provide good water vole habitat, and are known to be occupied (Figure 4.4). These water bodies are protected from flooding by unbreached flood defence embankments, and are only a short distance from some of the floodplain water bodies where water voles are present. It is therefore possible that these sheltered pools and ditches with more stable water levels may provide temporary refuges for water voles when water levels within the floodplain area are high and variable (Strachan et al. 2011). The man-made island in the vicinity of the water vole population on the north side of the river (Figure 4.1) could perform a similar function as a refuge (Strachan et al. 2011). Future design of floodplains for water vole conservation should consider the likely timing of flood events. Previous studies have reported negative impacts of winter (Macpherson et al. 2003) and spring (Moorhouse et al. 2009) flood events, although at Fishlake large floods occurred without any noticeable impact in both the summer and winter. However, as a general rule it could be expected that any negative impacts of flooding should be greatest in the autumn, when it is vital for juveniles to attain a certain weight (Strachan and Jefferies 1993), and in the winter, when water voles are less active (Leuze 1976).

4.6. Conclusions

The restored floodplain at Fishlake provides suitable habitat for water voles, and habitat preferences within the floodplain are comparable to those observed in rivers and streams. There is the potential for flooding to disrupt water vole populations, but in this case there was no observed negative impact following floodplain reconnection. Management of floodplain wetlands for water voles should aim to provide wide water bodies with, tall, diverse vegetation. It may also be appropriate to provide adjacent, non-flooding water bodies and islands as refuge areas during flood events, and to limit the extent of flooding at times of the year when water voles may be more vulnerable.

5. The aesthetic impact of dragonflies and debris on recreational experiences in a floodplain wetland

This chapter investigates the ways that human behaviour can be managed to impact the ecosystem services that people receive, specifically, the quality of their recreational experiences. Recreational experiences are a key ecosystem service that floodplain wetlands can provide, particularly in urbanised and agricultural floodplains where opportunities for people to interact with nature are rare. Improving and encouraging public access to the floodplain were specific targets of the Fishlake restoration project. This study used data collected in part of the Fishlake floodplain to analyse the impacts of two habitat components (odonates and debris) on recreational experiences. Odonates and debris are commonly found in floodplain habitats, and both of these components are likely to be affected by hydrological reconnection. Odonate larvae require permanent standing water, and the adults require wetland environments and their associated vegetation. Restoring hydrological connectivity is likely to increase the area of these habitats in a floodplain. Debris items are commonly carried in rivers (Figure 5.1) so can be deposited on floodplains by floods. Increasing the frequency of flood events is likely to increase the deposition of debris.

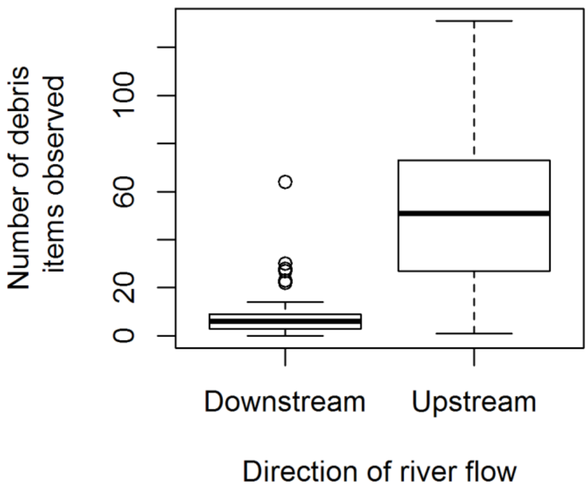

Figure 5.1. Debris items observed in five minute periods in the River Don at Fishlake on 58 occasions during 2011-2013. At Fishlake, flooding is largely driven by tidal influences, so most flood events occur when the river is backing up due to a tidal surge, thus appearing to flow upstream. The number of debris items in the river during upstream flow is significantly higher than during downstream flow (see Appendix C.1 for methods and results of the statistical test). Box and whisker plots show the median (bold line), interquartile range (box), and range within 1.5 × IQR (whiskers). Outliers are shown as circles.

This study analysed the net impact of the odonates and debris items that people noticed on their recreational experience. The study modelled spatial variation in the quality of these experiences at a fine resolution. This understanding could be applied practically to inform the design of footpath routes or wildlife viewing sites, and the framework presented could be generalised to other systems. This study has been prepared for submission to the Journal of Environmental Management, and is presented in this thesis in the form of the manuscript. The sections have been re-numbered and text has been formatted for consistency with the thesis, and all references can be found in the general references section. The dataset used in this chapter; including odonate abundances, debris quantities, and environmental explanatory variables, can be found in Appendix F.3 on the accompanying CD.

5.1. Abstract

Recreational experiences in natural environments are largely affected by the habitat components that people interact with, for example the physical features and organisms that they see. This interaction between people and habitat components is rarely considered in analyses of recreational experiences. In this study we consider a three-stage framework that describes the interaction between habitats and people. This framework considers the distribution of habitat components in the environment, the proportion of the available components that visitors notice, and the net impact of multiple components on the quality of the recreational experience. A recreational experience model based on these three factors was created for a case study lowland river floodplain, and was used to estimate visitor exposure to a combination of positive habitat components (dragonflies and damselflies) and negative components (natural and man-made debris). The experience model provided an index of net experience quality that showed spatial variation across the study floodplain. This analysis highlighted areas that would deliver positive experiences to visitors, in such detail that the results could inform the design of footpath routes, signage, or wildlife viewing sites. The experience modelling framework presented in this study could be applied to different habitat types and larger numbers of habitat components. The results of a sensitivity analysis indicated that the interaction between people and habitat components was a key part of the experience process, and that neglecting the noticeability (observation rate) of habitat components during modelling resulted in considerably different patterns in the spatial distribution of experiences.

5.2. Highlights

- A model of recreational experience quality was developed in a floodplain.
- The distribution of odonates and debris, their noticeability to visitors, and visitor preferences for these components, was modelled to calculate a net recreational balance.
- The model was used to predict spatial variation in recreational experience quality that could be used to design footpaths or wildlife viewing sites.
- The noticeability of odonates and debris was the most important component of the model; removing this component greatly influenced the management recommendations.

5.3. Introduction

Recreational experiences are a key ecosystem service that habitats can provide (Plieninger et al. 2013), so it is important to understand the impacts of habitat management on these experiences (Christie et al. 2007; McCool 2008). Previous studies of recreational ecosystem services have focused on identifying components of habitats, such as organisms and physical features that people desire to experience (Westerberg et al. 2010). An understanding of people's desired experiences can inform environmental management objectives at a general level (Bullock et al. 1998; Christie et al. 2007; Smyth et al. 2009), but does not necessarily allow decision makers to determine the best way to implement these objectives. For example, preference studies can be used to make recommendations that might improve recreational experiences, such as increasing the number of species that visitors see (Naidoo and Adamowicz 2005), or improving visitor access to facilities (Christie et al. 2007). However, to design on-the-ground management interventions that deliver these aims, a more detailed understanding is required; habitat managers must know how to increase visitor exposure to species, or access to facilities. It is therefore important to understand how visitors interact with components of a habitat, and how this interaction affects their experience of it.

Visitor recreational experiences can be affected by a range of factors, but in this study we focus on the impacts that physical and ecological habitat components can have on experience quality. We analysed these impacts using a three stage framework that describes the delivery of recreational experiences from habitat components to people. First, the potential experience that a habitat could provide is determined by the physical and biological components that are present there (Haines-Young and Potschin 2007; Bastian et al. 2012). Second, a visitor will only experience a proportion of this potential, depending on the area that they visit, the timing and duration of their visit, their awareness of the habitat and the components that might be present in it, and the relative crypsis of the components that are present (Hull and Stewart 1995; Hughes and Newsome 2005; Naidoo and Adamowicz 2005). Third, particular habitat components will impact visitor experiences differently. Some will be positive (i.e. will enhance the quality of the experience), and some will be negative (i.e. will reduce it), and the net balance of all components that are noticed by the visitor will determine the quality of the experience provided (Chenoweth and Gobster 1990; Bullock et al. 1998; Dorwart et al. 2009).

Typically, previous research has not considered all three factors in the experience process, and in particular has neglected the relationship between what is present in the environment, and what people notice. Research has focused on characterising visitor preferences for habitat components (Hanley et al. 1998; Hoehn et al. 2003; Birol and Cox 2007; Westerberg et al. 2010; Kenter et al. 2011), and has commonly used choice experiments to measure these preferences (Adamowicz et al. 1994; Hanley, Wright, et al. 1998). Some studies have integrated preference information with records of what people experience in the environment, for example through the use of on-site surveys, visitor employed photography (Dorwart et al. 2009; Nielsen et al. 2012), stakeholder mapping exercises (Fagerholm et al. 2012; Plieninger et al. 2013), or by integrating preference studies with field data recorded from the perspective of a visitor (Naidoo 2004; Naidoo and Adamowicz 2005). These methods can tell decision makers which habitat components people notice, and which are most desirable. However, they do not provide insights into the mechanisms that underlie the delivery of experiences from habitats to people. For example, in a study of forest recreational experiences it is not clear whether participants took more photographs of "negative" dead wood items than "positive" dead wood (Nielsen et al. 2012) because there were more examples present, because the examples were more noticeable, or because the items provoked a stronger participant response. To understand how manipulation of a habitat may affect recreational experiences, it is important to distinguish between the relative importance of ecological factors (e.g. total species richness) compared to human behavioural factors (e.g. trail routes, presence of guides, hide infrastructure) in affecting the visitor experience (Naidoo and Adamowicz 2005).

Previous studies have not integrated the three stages of the experience process; commonly because peoples recreational experiences have not been linked to what is present in the environment. This study demonstrates a framework for considering; (1) the relationships between ecological and physical factors and the distribution of two habitat components, (2) the noticeability of these habitat components to visitors, and (3) the relative preferences that people have for the habitat components. We apply this framework to model recreational experiences in a floodplain wetland case study. Floodplains are an important recreational resource (Gren et al. 1995), and are commonly managed to enhance their recreational potential. Among the habitat components that are often present in floodplains, we analysed one positive and one negative component. Odonates (dragonflies and damselflies) and debris items (including both natural and man-made debris) were chosen as examples of positive and negative habitat components respectively, because they were expected to have contrasting impacts on visitor experiences and were known to be consistently present at the study site. The experiences of the authors at the chosen study site and informal discussions with local residents indicated that debris items were generally perceived negatively by visitors, and the presence of odonates at the study site was known and popular. A dragonfly is used in the logo of a popular local walking route: the Thorne Round Walk. Additionally, odonates and debris do not typically move rapidly over large distances, and these components vary in their spatial distributions at a fine spatial scale (within metres).Visitor exposure to these habitat components was therefore likely to vary depending on their location in the study floodplain. These small-scale habitat components contrast with larger, more noticeable habitat components, such as cattle or wetland birds, which are likely to be equally noticeable from most points in the floodplain. Visitor exposure to larger components may however vary across larger spatial scales, such as between floodplain wetlands. Adult odonates are distinctive wetland organisms (Brooks and Lewington 1997), and are attractive and popular, both with wildlife enthusiasts and in wider culture (Simaika and Samways 2008; Lemelin 2007; Lemelin 2009). Debris accumulation is common in lowland floodplains because buoyant items are brought in by flooding (Williams and Simmons 1999). Both natural (e.g. wood or vegetation) and man-made (e.g. food or drink containers) debris items are known to negatively impact the visitor experience in coastal (Tudor and Williams 2003) and riverine (Williams and Simmons 1999) habitats.

Management of recreational experiences can involve manipulating the spatial location of visitors to alter their interaction with habitats, for example by altering footpath routes, adding boardwalks, or building wildlife viewing platforms (Orams 1996; Reynolds and Braithwaite 2001). In this study we modelled spatial variation in the quality of recreational experiences delivered by odonates and debris, to inform the design of a wildlife viewing site. We modelled recreational experience quality by applying the three-stage framework described above, using field and online survey data. The recreational experience model predicted the relative quality of visitor experiences at different parts of the study floodplain. This information was used to compare spatial variation in experiences, with the aim of informing the design of footpath routes or the placing of wildlife viewing sites. To assess the importance of considering habitat component noticeability when analysing recreational experiences, we conducted a sensitivity analysis to analyse the relative importance of the three stages in the framework in contributing to the accuracy of the recreational quality estimates.

5.4. Methods

5.4.1. Study site and chosen habitat components

Recreational experiences were analysed at a case study lowland river floodplain. The study site is located at Fishlake, near Doncaster in the United Kingdom (Figure 5.2a; Latitude: 53.611239, Longitude: -1.002889). The curvilinear site is bounded by the River Don to the south and a combined footpath and flood defence bank to the north, and receives inundation from the river through an engineered bank breach (Figure 5.2b). The habitat in the study area is a mosaic of open water, marsh, and wet grassland. The standing water provides habitats for aquatic organisms, including dragonflies and damselflies (Odonata), while the periodical flood events bring carry debris items from the river and deposit them across the floodplain.

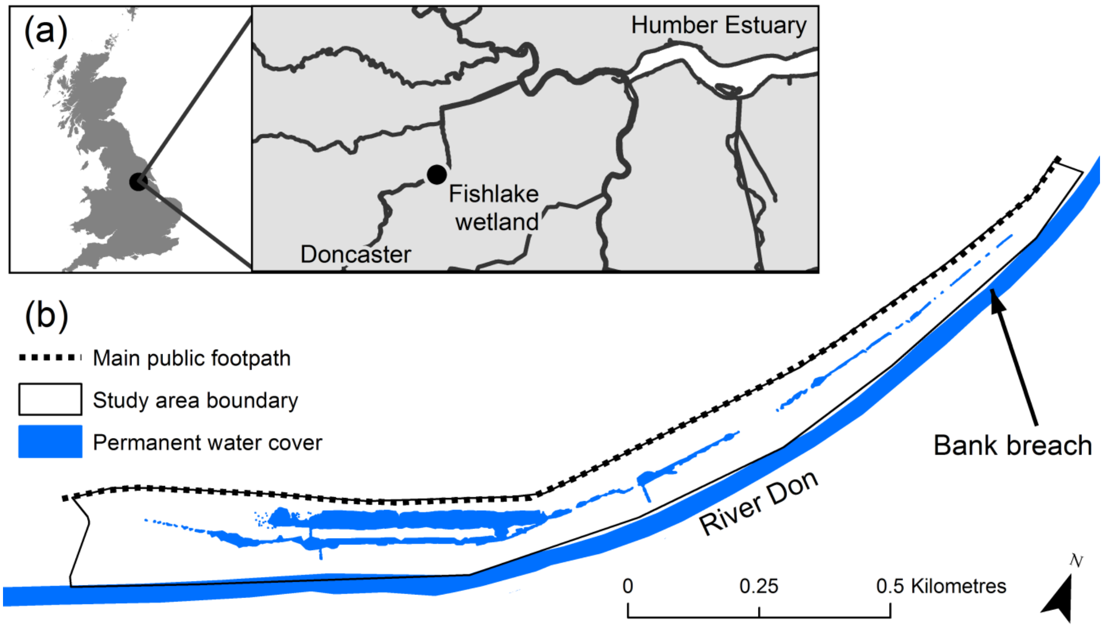

Figure 5.2. Overview of study site. (a) Location of Fishlake wetland in the Doncaster region and United Kingdom. (b) Permanent water cover and location of footpath at the floodplain study area.

5.4.2. Three stage modelling framework

5.4.2.1. Spatial distribution of odonates and debris

The abundance of odonates and debris items was measured in 100, 400 m^2 quadrats that were randomly located, but stratified across four habitat types across a gradient of flood exposure (referred to as “quadrat surveys”). Odonates were sampled at each quadrat on three days in August 2012 and three days during July-August 2013. Debris items (including both man-made and natural items) were counted on three occasions interspersed by periods of flooding: June 2012, August 2012 and May 2013. Searches for odonates were limited in duration to one minute.

Environmental data were mapped continuously over the floodplain area, so data were available for each of the quadrats. The following environmental variables were mapped; the mean proportion of time that the area within the quadrat was flooded (flood exposure), whether or not there was a permanent area of standing water inside the quadrat (water body permanence), an ordinal estimate of vegetation height (vegetation height), the minimum flow distance from the nearest flood entry point (flow distance), the distance from the shore at the highest recorded water level (distance from the high water mark), and topographic slope. Hydrological and topographic variables were derived from a detailed site topographic map and time series of water level measurements that spanned two years, using the methodology described in Richards et al. (in press). Vegetation height was mapped visually using 5 ordinal categories (0 cm, 1-20 cm, 21-40 cm, 41-75 cm, 76+ cm) (Hazel Stanworth, University of Sheffield; unpublished data). The flow distance variable was calculated using a least-cost pathway method (see Appendix D.1). To estimate the distribution of odonates and debris items across the whole floodplain, these data were used as explanatory variables to model the abundance of odonates and quantity of debris. The abundance of odonates and debris was modelled in response to different environmental predictor variables, depending on prior expectation of the environmental factors that might be important in determining their distribution. Odonate abundance was modelled in response to flood exposure, water body permanence and vegetation height. Debris quantity was modelled in response to flood exposure, flow distance, distance from the high water mark, vegetation height, and slope. Odonate abundance and quantity of debris were modelled as negative binomial responses using generalised linear mixed-effects models (GLMMs) (Skaug et al. 2006), and these models were simplified following a stepwise procedure to minimise Akaike's An Information Criterion (AIC) (Burnham and Anderson 2002; Crawley 2009). The resulting models of odonate abundance and debris quantity were then applied across the whole floodplain, to predict the distribution and density of odonates and debris across each of 825, 400 m^2 grid squares.

5.4.2.2. Observation rate of odonates and debris

At the same time as the surveys of each quadrat were conducted, the noticeability of odonates or debris to visitors was measured by surveys of each of the sample quadrats from the nearest point of the main public footpath ("remote surveys"). The number of odonates or debris items that were observed from the footpath was recorded. The footpath surveyor searched by eye and focused entirely on the survey quadrat in which the quadrat surveyor was present.

The data from the remote surveys were used, in combination with those from the quadrat surveys, to model the noticeability of odonates and debris. The noticeability of debris items was modelled as the proportion of items that were present in a quadrat that were observed from the footpath. This was modelled as a binomial response within a GLMM, using vegetation height, distance between the quadrat and the footpath, and quantity of debris present in the quadrat as explanatory variables (Skaug et al. 2006). The maximal model was then simplified stepwise using AIC as the criterion (Crawley 2007). The data from the remote surveys indicated that the observation rate of odonates attenuated rapidly with distance, to such an extent that it was practical to assume that the probability of observing an odonate from a distance greater than one quadrat was zero (see Results section 5.5.1). It was therefore assumed that a visitor within a quadrat would experience all of the odonates that were present within it.

The remote survey method may over-estimate the amount of debris noticed from the footpath by the average visitor, as it is likely that a member of the public would not observe the survey area in as much detail as the remote surveyor. However, the survey effort expended when sampling odonates and debris items was comparable, so this method can be applied to indicate the relative noticeability of these components. Alternative methods such as visitor surveys could have been applied to more realistically quantify the numbers of odonates and debris that were observed from the footpath. However, such methods would not have allowed the actual number of odonates and debris present to be measured, because it would not have been possible to simultaneously survey the entire area within the visitor’s line of sight. It would therefore not have been possible to quantify the observation rate of these components.

5.4.2.3. Public preferences for odonates and debris

The net experience provided by odonates and debris at a viewing site location was analysed using preference data from a choice experiment that was completed by The University of Sheffield staff and students. Choice experiments are commonly used to evaluate people's preferences for different habitat components (Adamowicz et al. 1994; Hanley, Wright, et al. 1998), as they allow preferences for individual components to be measured and compared quantitatively. Quantitative measurements of preferences are a flexible tool for prediction because the impacts of multiple components, and different combinations of components, can be assessed and reported in terms of a net preference. It would not be possible to precisely predict the net impact of a range of scenarios using qualitative measures of people’s preferences. Choice experiments are often used in economic analyses of recreational ecosystem services (Hanley, Wright, et al. 1998), but this approach can also be applied to compare preferences in either dimensionless or non-monetary units. Participants were presented with pairs of hypothetical scenarios based at the study floodplain and asked to choose the one which they would prefer to visit in each case. Each floodplain scenario varied in two characteristics: the quantity of debris present and the abundance of odonates present. Debris quantity was represented as a fixed-view photograph of part of the study site in which the quantity of debris had been experimentally manipulated. Factor levels for debris quantity were 0, 30, 50 and 150 debris items, and for each photograph a realistic mixture of natural and man-made debris was used (4:1 natural to man-made). Prior to beginning the survey participants were also shown some example photographs of debris aggregations at the study site. Odonate abundance was represented separately as a numerical value shown below each question image, alongside a greyscale drawing of a damselfly (see Figure 5.3 for an example choice set). A greyscale image was used to reduce the potential positive bias that may be encountered if a close-up colour photograph was used, as it is unlikely that a visitor in the field would experience odonates in such detail. To give context, visitors were shown an introductory video that presented individuals of a common odonate species at the site (*Ischnura elegans*) as they would likely be viewed in the field. Factor levels for odonate abundance were 0, 1, 3, 5, 7 and 10 individuals. A total of twenty-four choice sets were presented to the participants in two blocks, so each participant answered 12 questions.

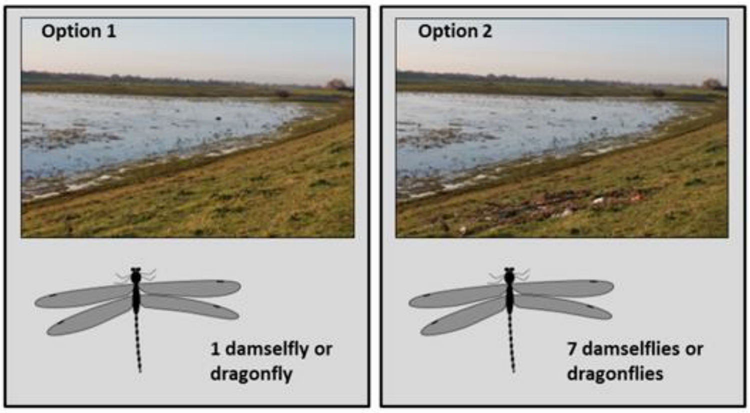

Figure 5.3. Example choice set used in the online survey. Participants were asked to select whether they preferred Option 1 or Option 2.

The relative preferences of participants for debris and odonates were modelled using a conditional logistic model (Therneau 2013). Debris quantity, odonate abundance, and a quadratic term for odonate abundance were used as explanatory variables. A quadratic term for odonate abundance was used because the recreational benefits provided by habitat components commonly do not scale linearly but show diminishing returns (Rambonilaza and Dachary-Bernard 2007). Relative preference was expressed using a willingness-to-pay approach, but the aim was to characterise the net value of recreation in terms of odonates and debris, rather than in monetary terms. It was expected that the recreational benefit provided by odonates would be offset by the negative "cost" of seeing debris. In this case, the recreational benefit provided by an odonate could be characterised as the number of debris items that a participant was willing to tolerate seeing, in order to see an odonate.

The number of debris items that a participant was willing to see in order to experience an odonate (referred to henceforth as the willingness to see debris) was calculated as the ratio –n/m where n was the estimated coefficient of odonate preference (as estimated by the conditional logit model), and m was the estimated coefficient of debris preference (Aizaki 2012). The negative n term was necessary to give a positive estimate of the number of debris items.

The willingness to see debris was used to calculate an "experience balance" for locations in the floodplain. The experience balance indicates the net visitor experience, in units of debris. A more positive experience balance indicates an excess of odonates and therefore a more positive net visitor experience, while a more negative balance indicates a less positive experience. To calculate the experience balance it was first necessary to know the maximum number of debris items that a visitor was willing to see, in order to see the number of odonates present in the grid square. This was calculated as:

$$W = o \times m_o + o^2 \times m_p$$

Where W was the willingness to see debris, o was the number of odonates present, m_o was the marginal willingness to see debris value for odonates, m_p was the marginal willingness to see debris value for the odonate quadratic term, and d was the quantity of debris present. To calculate the net balance, this willingness to see debris was compared to the number of debris items that the visitor actually saw. The debris balance was thus quantified as:

$$b = W - d$$

Where b was debris balance, W was the willingness to see debris items, and d was the quantity of debris present.

The presence of odonates and debris items is temporally and spatially variable, so stochasticity was incorporated into the recreational experience model through a bootstrap method. For each bootstrap replicate, the odonate and debris abundance

GLMMs were bootstrapped using a model-based (fixed –X) technique, with resampling stratified across survey dates (Fox 2008). This created variability in the model coefficients, and therefore in the estimate of odonates and debris items on the floodplain. Stochasticity was not incorporated into the noticeability models because it was not possible to quantify temporal variability in odonate noticeability, so noticeability for both odonates and debris was assumed to be fixed across bootstrap replicates. The number of odonates and debris items present, and the number noticed at each viewing location were calculated for each of 300 bootstrap replicates, and the mean values of the 300 replicates were used to calculate debris balances when comparing different viewing sites.

5.4.3. Sensitivity analysis: the importance of considering all three stages in the experience process

The series of statistical models described above were combined to form a model that was used to simulate the relative quality of recreational experiences in different parts of the floodplain. This model incorporated all three stages in the experience process, and is referred to throughout as the total model. To compare the relative importance of each of the three stages in the experience process, three additional indicators that included only parts of the total model were calculated. The first indicator only considered the spatial distribution of positive and negative components, and was quantified as the ratio of odonates to debris (odonates / debris) that were present within the same quadrat as the visitor (the distribution model). The second indicator of recreational experience combined distribution data with the relative preferences of visitors for odonates and debris, and was quantified as the debris balance of the odonates and debris that were present in the same quadrat as the visitor (the distribution and preference model). The final indicator considered the presence and noticeability of positive and negative components, but not the relative preferences of visitors for them. This final indicator was quantified as the ratio of odonates to debris (odonates / debris) that were noticed by the visitor (the distribution and noticeability model). These partial models of recreational experience were run over the same set of bootstrap replicates as the whole-experience model, so the same number of odonates and debris were always present on the simulated floodplain. The difference in relative performance between each of these indicators and the whole-experience indicator was analysed as the Spearman's rank correlation coefficient (Spearman's rho) between the two resulting debris balances. The statistical significance of this coefficient was not assessed because the number of wildlife viewing sites in the sample, and therefore the number of degrees of freedom available, was arbitrary.

5.5. Results

5.5.1. Distribution, noticeability, and preferences for odonates and debris items

Of the 100 quadrats surveyed, debris was present on at least one occasion at 95 quadrats, and odonates were present on at least one occasion at 79 quadrats. Odonate abundance was significantly greater in quadrats that had more frequent flood exposure, contained some permanent standing water, and that had taller vegetation (Table 5.1). The quantity of debris present within a quadrat decreased significantly with increasing flood exposure, vegetation height, and flow distance from the bank breach (Table 5.2). The majority of odonates (more than 95%) recorded within the quadrats were damselflies, most commonly *Ischnura elegans* and *Coenagrion puella*. Occasional *Calopteryx splendens*, and the dragonflies *Sympetrum striolatum* and *Anax imperator* were also observed. Approximately three quarters of the debris present in the quadrats was natural; mainly wood or riparian vegetation. The man-made debris that was observed comprised mainly plastic bottles, pieces of polystyrene or food wrappers.

Debris was observed from the footpath at least once at 87 of the 100 quadrats, and odonates were observed from the footpath at only two quadrats. The quantity of debris recorded as observed from the footpath was slightly greater than the actual quantity recorded within the quadrat on four occasions. This occurred at low levels of debris, and was likely due to misidentification of bare ground or floating foam as debris. In these instances the observed debris was recorded as the actual amount present in the quadrat, i.e. noticeability was recorded as perfect. The observation rate of debris increased significantly when there was a larger amount of debris present in a quadrat, and decreased significantly as the observer was further away (Table 5.3). It was not possible to model the noticeability of odonates statistically due to their very low observation rate, but odonates were unlikely to be visible from distances much greater than the 28 metre diagonal length of a quadrat. The two recorded odonate sightings occurred at quadrats that were three and 30 metres from the footpath, and were both sightings of *Anax imperator*, a large, rare, dragonfly species.

The choice experiment survey received responses from 308 people. The participants showed a significant negative preference for floodplain scenarios with greater debris quantities (Table 5.4), and preference for odonates showed a significant quadratic relationship with increasing odonate abundance (Table 5.4). The willingness to see debris showed a concave quadratic relationship, plateauing at around eight odonates (Figure 5.4). The observation of a single odonate was valued at 27 debris items (Figure 5.4), which corresponds to an aggregation covering approximately $2.5m^2$.

Table 5.1. GLMM of odonate abundance in floodplain quadrats.

	Estimate	Std. error	z	p
Intercept	-1.946	0.389	-4.99	< 0.001
Flood exposure	0.018	0.007	2.52	0.011
Water body permanence	0.964	0.244	3.94	< 0.001
Vegetation height	0.265	0.083	3.17	0.001

Table 5.2. GLMM of debris quantity in floodplain quadrats.

	Estimate	Std. error	z	p
Intercept	2.993	0.330	9.04	< 0.001
Flood exposure	-0.031	0.004	-6.47	< 0.001
Flow distance from flood entry point	-0.007	0.002	-2.51	0.012
Vegetation height	-0.130	0.063	-2.06	0.038
Distance from high water mark	0.009	0.005	1.77	0.076

Table 5.3. GLMM of noticeability of debris items from the bank.

	Estimate	Std. error	z	p
Intercept	2.344	3.366	0.69	0.48
Distance from footpath	-0.153	0.028	-5.34	< 0.001
Vegetation height	-0.133	0.081	-1.63	0.1
Number of debris items present	0.074	0.011	6.32	< 0.001

Table 5.4. Conditional logit model of relative preferences of survey participants for debris and odonates.

	Estimate	Std. error	z	p
Alternative specific constant	-0.43	0.058	-7.49	< 0.001
Debris quantity	-0.03	< 0.01	-29.40	< 0.001
Odonate abundance	0.87	0.04	19.93	< 0.001
Odonate abundance squared	-0.05	< 0.01	-12.56	< 0.001

5.5.2. Spatial variation in the estimated quality of recreational experiences

The simulations estimated the abundance of odonates and quantity of debris that were present across the whole study area, and using this method an average of 868 odonates and 6674 debris items were estimated to be present in the study floodplain during any one visit by a member of the public. The total model of recreational experiences estimated that the net debris balance experienced at the 150 locations would vary between approximately -57 and 27 items, with a mean of -21 items (negative numbers indicate net negative experiences, and positive numbers indicate net positive recreational experiences). The best recreational experiences were delivered in areas that were within or close to larger water bodies, and were distant from the flood entry point (Figure 5.5).

5.5.3. Sensitivity analysis

The performance of the partial models of recreational experience was assessed by comparison with the total model. The predictions from the distribution model were the most different from those of the total model (Spearman's rho = 0.62). The distribution and preference model also gave considerably different results to the total model (Spearman's rho = 0.69), and the predictions from the distribution and noticeability model best matched those made by the total model (Spearman's rho = 0.93).

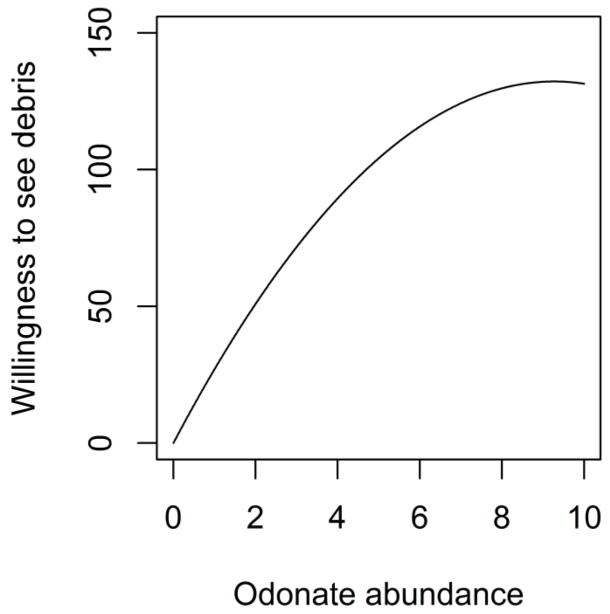

Figure 5.4. Willingness to see debris for increasing abundances of odonates (units are number of debris items that participants were willing to see).

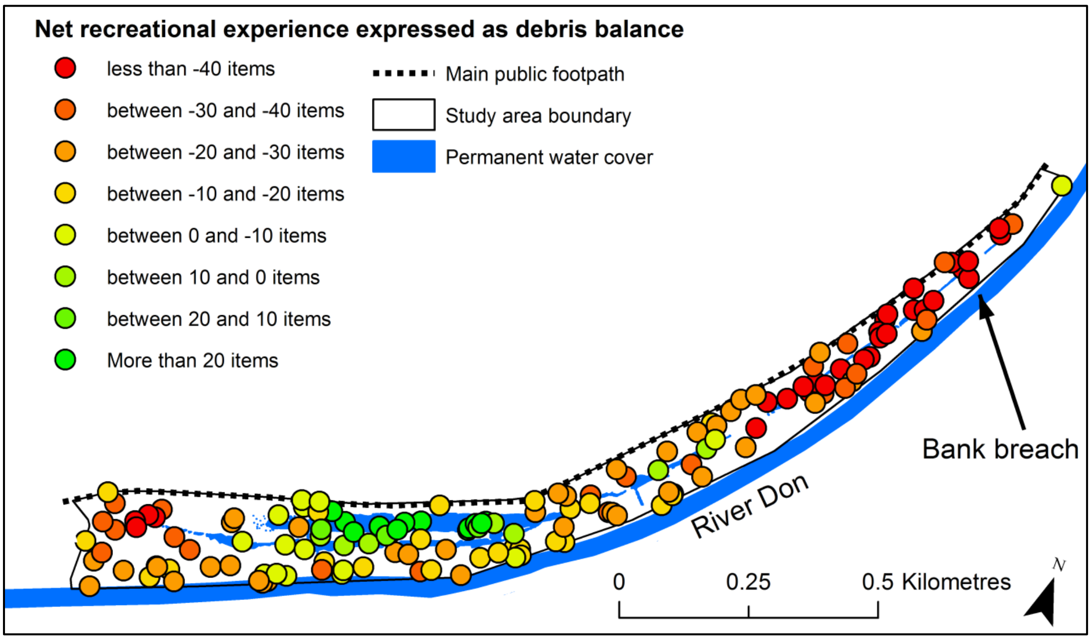

Figure 5.5. Net visitor recreational experience expressed as debris balance for each of 150 floodplain locations. More positive values indicate more positive recreational experiences.

5.6. Discussion

This study analysed the impacts that two habitat components can have on recreational experiences. It integrated an understanding of the physical and ecological factors that explained the distribution of odonates and debris, the noticeability of these components to visitors, and visitor preferences for the two components. This framework enabled the net recreational experience quality ,delivered by odonates and debris to be predicted at a high spatial resolution within the study floodplain. The net experience balance is not an absolute measure of recreational quality, but an index that allowed a number of management options to be compared objectively (Figure 5.5). In the case study floodplain, the most positive experiences were delivered in wetter areas which were further from the bank breach, because in these areas interaction with odonates was most likely, and exposure to debris was reduced. This detailed knowledge could inform management, as visitor behaviour could be manipulated to encourage interaction with optimal areas of the site (Orams 1996; Reynolds and Braithwaite 2001; McCool 2008), for example through the construction of a series of viewing platforms or alteration of the footpath route (Reynolds and Braithwaite 2001; Suh and Samways 2001). To make spatially detailed estimates of the recreational experiences provided by a habitat, an understanding of all three stages of the experience process is required. This study provides a novel framework for considering the interaction between people and the environment when analysing the provision of recreational experiences.

The noticeability of habitat components impacts visitor recreational experiences, because noticeability affects the interaction between visitors and the environment. Odonates were rarely observed from the footpath because of their small size and cryptic behaviour (Brooks and Lewington 1997; Taylor 2013), and as a result the interaction between visitors and odonates was limited. In contrast, debris items were noticeable from the footpath even at considerable distances, probably because they formed aggregations (Storrier et al. 2007) that contrasted in colour with the background vegetation (Bishop and Miller 2007). This differential observation rate had an impact on the habitat components that a visitor was likely to interact with, and therefore on the quality of recreational experiences.

The partial models of recreational experience that did not consider the noticeability of habitat components (the distribution model, and the distribution and preference model) gave very different results when compared to the total model of experience. This suggests that the different observation rate of odonates and debris had a relatively large impact on visitor recreational experiences. When the noticeability of habitat components is not considered explicitly during analysis then it is assumed to be equal, and in the case of the distribution and preference model it was assumed that the noticeability of both odonates and debris was very low. This assumption resulted in an underestimate of the quantity of debris that visitors were exposed to. The error in estimating recreational experience quality based on habitat component distribution data and preference information indicates that there is a risk in combining preference data with habitat descriptions of sites or regions, as is commonly done in benefit transfer valuations of ecosystem services (Troy and Wilson 2006; Liu et al. 2010). Valuations derived through applications of benefit transfer that use the occurrence, rather than experience, of habitat components may not accurately represent the true recreational value because while habitat components have some existence value (Sutherland and Walsh 1985; Pate and Loomis 1997), much of their recreational value comes from being directly experienced (Green and Elmberg 2014). Benefit transfer studies should ensure that they use data that represent the likely experiences of actual visitors, rather than the potential of the components present in the habitat, in order to minimise the chance of under- or overestimating recreational value. It may be possible to ignore noticeability when comparing habitat components that are similarly noticeable to visitors, because the ratio of what is present will be similar to the ratio of what people will experience. However, cases like the example presented here, where the noticeability of habitat components is very different, are likely to be common.

Although the combined distribution and noticeability model of recreational experience performed well in relation to the total model, there were subtle differences in its predictions. These differences were a result of the nonlinear relationship between public preferences for these components (Figure 5.4). When the preference relationship between positive and negative habitat components is nonlinear, the actual visitor experience must be estimated as accurately as possible, because the absolute quantity of the habitat components will affect the balance between them. For example, the net debris balance if one odonate and one item of debris were experienced would be 26. This is very different to the net balance if 10 odonates and debris items were experienced, which would be 121. The net recreational impact of odonates and debris would thus be different; even though the ratio of odonates to debris in each case would be the same. Such nonlinear preference relationships are likely to be common between different habitat components because many positive habitat components have a novelty value (Moscardo and Saltzer 1993; Hughes and Newsome 2005), so increasing numbers of them can give diminishing aesthetic returns (Rambonilaza and Dachary-Bernard 2007). There are additional advantages to using a preference relationship to estimate the net recreational experience when comparing more than two habitat components. The relative utility of different habitat components, and corresponding willingness to see negative components, can be used to summarise the net impact of multiple, positive and negative habitat features as an experience balance. This method can additionally be used to calculate an economic value for the habitat or habitat improvements if desired (Carson and Mitchell 1993; Hanley et al. 1998; Westerberg et al. 2010).

The three stage framework outlined in this study does not represent the entire complexity of recreational experiences, such as the longer-term emotional, behavioural and cultural impacts of experiences in nature (Kaltenborn 1997; Ballantyne et al. 2011). However, the net recreational balance of habitat components provides an objective indicator of recreational quality that is suitable for application to habitat management decision making at local spatial scales (within management units). The framework applied in this case study is flexible and could be applied to management problems that consider larger numbers of habitat components in any habitat type, or that involve physical habitat modification. Additionally, variation in awareness or preferences for habitat components could be incorporated, to assess the recreational experiences of different socioeconomic (Birol et al. 2006) or visitor groups (Kenter et al. 2013). A potential constraint with this approach is that it required a large amount of data, as it combined a traditional choice experiment with field sampling that was more intensive than a visitor participation exercise. However, future applications of the three stage framework may be able to utilise existing data. The habitat preferences of some taxa that are of interest to the public have previously been described in detail (Buckton and Ormerod 2010; Besnard et al. 2013), and at many sites the distributions of key species are known (Ross-Smith et al. 2011). The noticeability of organisms and other landscape features is relatively straightforward to quantify through field surveys, and when data collection is not feasible it is possible to make assumptions about noticeability, as were made for odonates in this study. At larger spatial scales, interaction of visitors with larger habitat or landscape components such as mountains or woodlands can be estimated using viewshed analyses (van der Horst 2006). Quantification of the relative preferences of the public for a range of habitat components is a considerable challenge, as designing and conducting choice experiments, particularly for complex designs involving large numbers of habitat components, requires a large participant base and can be time consuming (Johnson et al. 2013). However, such choice experiments have previously been conducted in a range of habitat types including wetlands (Hoehn et al. 2003; Westerberg et al. 2010), tropical islands (Kenter et al. 2011), and temperate forests (Hanley et al. 1998), so applicable preference data may already be available in the literature. Alternatively, other indices of popularity, such as Google search volume (Żmihorski et al. 2012) taxa rarity (Tournant et al. 2012), or qualitative methods such as focus groups (Moran et al. 2007) and expert knowledge (Strager and Rosenberger 2006) could be used to estimate the relative preference of the public for habitat components.

5.7. Conclusions

Environmental management for the purpose of recreation should not only maintain and protect habitat components, but also provide opportunities for visitors to interact with them (McCool 2008). To predict recreational experiences and inform management at fine spatial scales it is necessary to understand this interaction between visitors and habitat components. The noticeability of different habitat components plays a key role in determining the subset of available components that visitors interact with, and therefore also their net recreational experiences. Noticeability of habitat components should therefore be considered during analyses of recreational experiences, and similarly during recreational ecosystem service valuations. The three stage framework for analysing recreational experiences presented in this paper considered the noticeability of habitat components to visitors, and allowed detailed estimates of the net visitor experience to be made in relation to two habitat components. These estimates were made at a sufficiently detailed spatial scale to allow on-site management planning, such as the design of footpath routes or signage to create wildlife viewing areas. Future studies of recreational experiences should consider the noticeability of habitat components, and could utilise the modelling framework outlined in this study. This framework is applicable to other habitat types and management problems, and could be extended to include more complex combinations of habitat components and variability among visitors.

6. Floodplain heterogeneity increases habitat multifunctionality but reduces the optimality of ecosystem service provision

In this chapter, information on multiple ecosystem services is synthesised to investigate the ways that habitat management may affect a whole suite. Consideration of multiple benefits, and the trade-offs between their provision, is a core aim of the ecosystem services approach. This study integrates ecosystem service indicators that were considered in Chapter 4 and 5 with additional data on flood storage volume, and the habitat preferences of wetland birds and cattle. These indicators are components of ecosystem services which are important at the Fishlake floodplain.

This study focused on hydrological heterogeneity as a characteristic of floodplain habitat structure that can be altered by management. It simulated a range of hypothetical floodplain management scenarios using data collected at Fishlake. The simulation was used to analyse the relationships between hydrological heterogeneity and two characteristics of the suite of ecosystem services that could be provided; the optimality of provision and the multifunctionality of the habitat. This study has been prepared for submission to the Journal of Applied Ecology, and is presented in this thesis in the form of the manuscript. The sections have been re-numbered and text has been formatted for consistency with the thesis, and all references can be found in the general references section. The plant community dataset used in this chapter can be found in Appendix F.4, and the ecosystem service indicator and environmental data can be found in Appendix F.5, both on the accompanying CD.

6.1. Summary

1. It is important to understand how aspects of habitat structure, such as habitat heterogeneity, can affect the ecosystem services that a habitat provides. Information on multiple ecosystem services can be synthesised to describe the general characteristics of a suite of services, such as the extent to which service provision is optimal, or a habitat is multifunctional. This study investigated the relationships between habitat heterogeneity and these two characteristics of ecosystem services, using a floodplain case study.

2. The optimality and multifunctionality of floodplain mosaics were analysed using a random sampling simulation. Floodplain habitat patches that varied in flood exposure were sampled for seven ecosystem services, and the relationships between flood exposure and service provision were modelled. The sampled floodplain patches were combined at random to simulate larger floodplain mosaics, and the provision of five ecosystem services was estimated for each simulated floodplain.

3. The simulated floodplain mosaics were optimised by iteratively replacing some of their component patches. The habitat heterogeneity and closeness to optimality of the simulated floodplains was correlated, and the habitat heterogeneity and multifunctionality of the optimal simulated floodplains was separately correlated. Separate analyses were carried out for the ten pairwise trade-offs between ecosystem services, as well as for an optimisation of all five services simultaneously.

4. The sampled ecosystem services showed different responses to flood exposure, and there was a general trade-offs between two bundles of services; dry areas supported beef cattle and grassland biodiversity, while wetter areas supported wetland biodiversity and provided recreational interest. In general, the most optimal suites of services were provided by more homogenous floodplain mosaics. Amongst the optimal services however, those that were more multifunctional were also more heterogeneous.

5. Optimal floodplains were relatively homogenous because individual ecosystem services were provided best within specific environmental ranges, and because their provision scaled positively with increasing habitat area. Ecosystem service provision was thus maximised when large areas of the same types of habitat were present. Some of the more heterogeneous floodplains were more multifunctional because different ecosystem services were provided by different habitat types.

6. Synthesis and applications: High spatial heterogeneity in habitat conditions is a characteristic feature of floodplains, and is commonly desirable for management. This strategy is likely to provide multifunctional suites of services, but runs the risk of reducing the optimality of the floodplain. It is possible to provide suites of services that are both optimal and multifunctional, but an understanding of the specific environmental requirements of the different ecosystem services is required.

6.2. Introduction

The concept of ecosystem services can provide a framework for informing the design of habitat management (Tallis and Polasky 2009; Fish 2011). An ecosystem service framework should describe the effects of management on the provision of multiple services (Tallis and Polasky 2009), and allow decision makers to predict the outcomes of different management scenarios (Nelson et al. 2009). Such a framework can be used to inform decision making at different scales of complexity; for example, it can enable habitat managers to target the production of particular ecosystem services (Johnson and Curtis 2001), or help them decide how to balance a trade-off between two services (Hansson et al. 2005). More generally, an ecosystem service framework could be applied to synthesise information on multiple ecosystem services and describe the characteristics of a whole suite. This study considers the effects of habitat management on two general characteristics of ecosystem services; the optimality of service provision, and the multifunctionality of a habitat.

This study considers Pareto optimality, which is commonly utilised for decision-making in multiple-criteria problems in economics and engineering (Ngatchou et al. 2005). In a habitat management decision there are a number of possible management scenarios, each of which will have a different outcome in terms of its provision of multiple ecosystem services. Pareto optimality is defined by comparing this range of possible management scenarios against each other, rather than by applying criteria or targets that are set by decision makers. Pareto optimality is therefore an objective and dimensionless measure of optimality, which does not depend on the weighting that the stakeholders or decision makers place on the different ecosystem services.

The suite of ecosystem services that a habitat provides is defined as optimal if the provision of one service cannot be improved upon without degrading the provision of the others (Ngatchou et al. 2005). In the case of two ecosystem services, Pareto optimality can be identified by comparing scenarios that provide the same level of one ecosystem service. If a number of scenarios are equal with respect to the first service, the Pareto optimal solution is the one which provides the highest level of the other service, because no matter which ecosystem service an individual prefers, this option would provide the greatest all-round benefit. All of the possible management scenarios can be visualised in two dimensions by plotting the predicted value of Service 1 against the predicted value of Service 2. This visualises the “decision space” (Reed et al. 2013) (see Figure 6.1 for a conceptual diagram). In the decision space, the range of Pareto optimal management options form the boundary, or Pareto frontier (black dashed line on Figure 6.1). Any management option that falls within the Pareto frontier (inside the light grey decision space on Figure 6.1) is suboptimal because there is no logical reason for a decision maker to select it; there are options on the Pareto frontier that could provide a higher level of Service 1 without degrading Service 2, or vice versa. There are commonly multiple Pareto-optimal management scenarios that deliver optimality in different ways. For example, an optimal scenario could maximise the provision of one ecosystem service to the detriment of the other, or could provide a more even balance of the two services. The degree of evenness between the provision of multiple ecosystem services can be considered as the relative multifunctionality of provision.

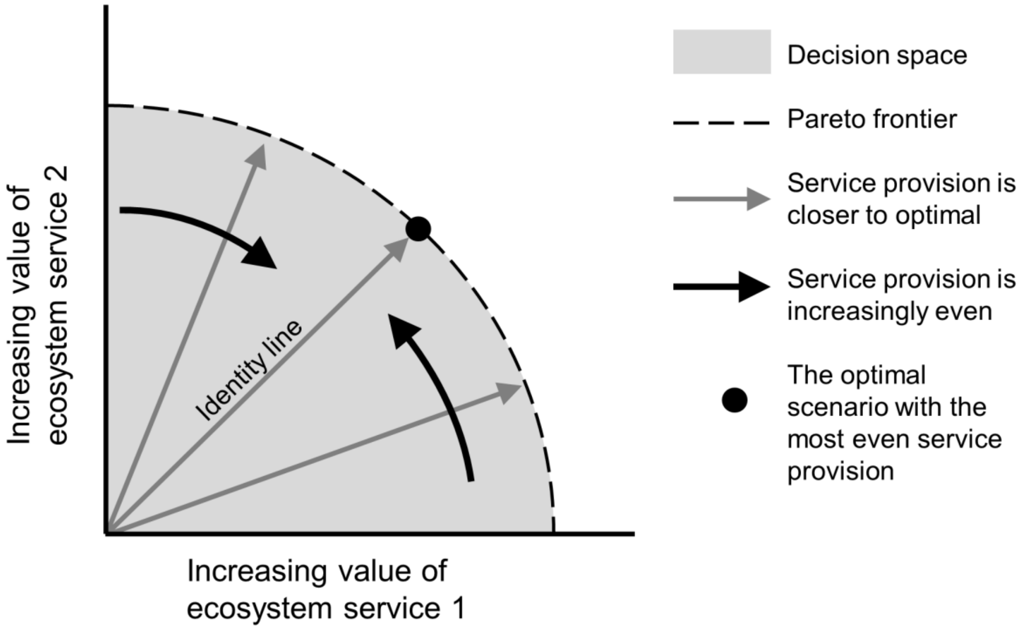

Figure 6.1. Conceptual diagram indicating a trade-off between two ecosystem services. The decision space of possible management outcomes is shown, bounded by the Pareto frontier of optimality. Any point within the decision space is suboptimal. Service provision is closer to optimal for scenarios that are closer to the Pareto frontier. Service provision is increasingly evenly balanced between the two services for scenarios that are closer to the identity line, and these scenarios are thus defined as more multifunctional.

There is increasing interest in managing habitats multifunctionally (Wiggering et al. 2006). Multifunctional habitats provide a balance of multiple ecosystem services, while monofunctional habitats provide one service at the expense of others (Otte et al. 2007). This categorical definition of multifunctionality can be extended to a continuous definition. Multifunctional habitats are commonly desired because they evenly balance the provision of multiple ecosystem services (Rouquette et al. 2011), so scenarios that provide a more even balance of multiple services may be defined as more multifunctional. In Figure 6.1, monofunctional habitat scenarios would be plotted along the axes, while multifunctional habitats that provide even suites of services would be plotted closer to the identity line. This index of multifunctionality makes the assumption that the values for both services are normalised, and that they are weighted equally. This is unlikely to be the case in the majority of habitat management decisions, but in the absence of weighting information we assume that this is the case for the purpose of this study. Differential weighting of ecosystem services could be achieved in future by scaling the services differently before calculating the overall evenness of provision. The black arrows in Figure 6.1 thus indicate dimensions of increasing multifunctionality.

Habitat multifunctionality and ecosystem service optimality are not mutually exclusive. There are likely to be a number of optimal management scenarios which lie along the Pareto frontier, and these scenarios will vary in their multifunctionality. Similarly, there are likely to be multiple scenarios that provide highly even suites of services, and these scenarios will vary in their distance from the Pareto frontier. The black circle in Figure 6.1 indicates a scenario that would simultaneously be multifunctional and would provide an optimal suite of ecosystem services.

The ecosystem services that a habitat provides depend on the way that it is managed, and previous studies have compared multiple management scenarios to identify management practices that could deliver habitat multifunctionality (Rouquette et al. 2011) or optimal suites of services (Sanon et al. 2012; Lautenbach et al. 2013). The recommendations that these studies make are typically specific to particular sites, habitat types, and groups of ecosystem services (Rouquette et al. 2011; Sanon et al. 2012; Lautenbach et al. 2013), so are limited in their generality. To transfer knowledge between systems it is necessary to relate optimality and multifunctionality to general patterns in habitat management, such as characteristics of the habitat structure. For example, habitat heterogeneity is a general characteristic of habitat structure that can be defined for any habitat type. This study focuses on habitat heterogeneity because of its known ecological importance in affecting biodiversity (Ward et al. 1999), and because a link between heterogeneity and multifunctionality has been previously suggested (Brandt 2003; Mander et al. 2007).

The provision of ecosystem services depends on environmental conditions (Morris et al. 2009). Different habitat types have different environmental conditions, so can provide different ecosystem services (Maskell et al. 2013). Habitat heterogeneity may therefore provide multifunctionality by allowing the provision of multiple services to be segregated across different habitat types within a mosaic (Mander et al. 2007). The impacts of habitat heterogeneity on ecosystem service optimality are more difficult to predict. Many ecosystem services, such as agricultural production, are likely to be maximised within a narrow range of environmental conditions (Letey 1985). Optimal habitats that are monofunctional for these services are therefore likely to be highly homogenous, as is commonly the case in intensive agricultural regions (Roschewitz et al. 2005). Optimal habitats that are not monofunctional may be more heterogeneous, but their heterogeneity is likely to depend on the number of ecosystem services considered, and the spatial scaling relationships of each service.

In this study the relationships between the heterogeneity of habitat mosaics, their multifunctionality, and the optimality of their ecosystem service provision, were investigated in a lowland river floodplain. Floodplains provide a wide range of ecosystem services (Tockner and Stanford 2002; Posthumus et al. 2010), and previous studies have investigated the impacts of floodplain management on multifunctionality (Rouquette et al. 2011) and service optimality (Sanon et al. 2012). The provision of floodplain ecosystem services is closely linked to the hydrological conditions that are present (Morris et al. 2009; Rouquette et al. 2011), and heterogeneity in hydrological conditions is a characteristic feature of floodplain habitats that can be altered by habitat management practices (Ward et al. 1999).

The aim of this study was to investigate the relationship between habitat heterogeneity and the suite of ecosystem services that a floodplain mosaic can provide. In particular, we analysed the optimality of the suite of ecosystem services provided, and the multifunctionality of the habitat. We first analysed variation in ecosystem service provision across a hydrological gradient, by sampling seven ecosystem service indicators in quadrats of varying flood exposure. We analysed indicators of cattle production, flood storage, three indicators of recreational value, and two indicators of different aspects of plant biodiversity. A series of resampling simulations were then performed to investigate the different ways that the sampled habitat patches could be put together to form larger mosaics. The performance of the simulated mosaics was analysed in relation to a subset of five ecosystem service indicators. Only five of the original ecosystem service indicators were used for this analysis because two of them; debris occurrence and flood storage volume, were strongly affected by the spatial configuration of the habitat mosaic, which it was not possible to model. To investigate the relationships between habitat composition and the optimality of ecosystem service provision we performed optimisation exercises to find optimal management scenarios for each pair of the five ecosystem service indicators separately, as well as for all five indicators together. For each comparison we analysed the relationship between hydrological heterogeneity and the closeness of the scenarios to ecosystem service optimality. For the optimal floodplain scenarios, we then correlated hydrological heterogeneity against habitat multifunctionality.

6.3. Methods

6.3.1. Study site and selection of ecosystem service indicators

Floodplain ecosystem service indicators were sampled across a gradient of flood exposure at a periodically inundated floodplain of the River Don, at Fishlake in the United Kingdom (Latitude: 53.611239, Longitude: -1.002889). Although formerly an active floodplain, the study site was drained historically and the river was embanked in the 1600s. The site has been used primarily as a flood storage area since the 1940s (flooding on average twice per year) and has also been used for beef production over this period. In August of 2009, breaches were made in the river embankments to allow a more natural flooding regime, under which the area is inundated more frequently (approximately 130 times per year), to a varying degree on each occasion. The primary aim of this change was to restore the floodplain habitat, and consequently the biodiversity that is characteristic of such systems. However, as with all habitats, the site supplies various ecosystem services. The Fishlake floodplain therefore provides an opportunity to analyse the relationships between hydrological conditions and the provision of multiple services.

We analysed three recreational ecosystem service indicators, one provisioning service indicator, one regulatory service indicator, and two biodiversity indicators. The selected service indicators were chosen to cover a range of the categories described in the Millenium Ecosystem Assessment (MA 2005), and because of their relevance to the study site and the objectives of the Fishlake floodplain restoration project. Flood storage capacity was measured as the regulatory ecosystem service indicator, and the presence of cattle dung was the indicator of the suitability for beef cattle production; a provisioning service. The recreational indicators were (1) the presence of wetland birds, (2) the presence of odonates (dragonflies or damselies), and (3) the presence of debris items (any man-made items or any natural items such as tree branches that were judged to be foreign to the floodplain). The two biodiversity indicators were (1) the similarity of the plant community to a National biodiversity target (floodplain grazing marsh), and (2) the number of regionally important wetland plant species that were present (species listed under the local Biodiversity Action Plan).

6.3.2. Field datasets

One hundred quadrats, each 400 m^2 in area, were sampled for the seven ecosystem service indicators and five additional environmental variables; flood exposure, distance from the flood entry point, slope, altitude, and vegetation height. The quadrats were selected from a grid by random stratified sampling. There were four equally-sized strata across a gradient of flood exposure, from areas that were permanently wet to those that were almost permanently dry (see Appendix D.1 for a map of quadrat locations).

Plant biodiversity is categorised as an ecosystem service in this study because species and communities have an intrinsic cultural value (Chapin et al. 2000) which has been recognised through legislation or guidance; in this case through regional species action plans and national priority habitat status (Natural England 2013). Plant communities in each quadrat were surveyed in June or July in 2012. All higher plant taxa present within each quadrat were identified and assigned a frequency score on the DAFOR scale (Brodie 1985). Taxa were identified to species level except in the case of the grasses *Agrostis* and *Poa*, which were identified to genus level, and filamentous algae were additionally classified as a group. Two indicators of biodiversity value were calculated for each plant community (one at a regional scale, one national indicator). The value of each plant community in contributing to the biodiversity of the region was quantified as the number of plant species that were listed on the Local Authority biodiversity action plan (henceforth, BAP species richness) (Appendix D.2). The value of each plant community in contributing to national biodiversity was quantified as its similarity to a nationally important habitat type; floodplain grazing marsh. Floodplain grazing marshes are a United Kingdom biodiversity action plan priority habitat (Mountford et al. 2006), and there is government and research interest in conserving and restoring these communities (Mountford et al. 2006). Similarity to floodplain grazing marsh was defined as the proportion of floodplain grazing marsh species present in a sampled community (henceforth, FGM similarity). Floodplain grazing marsh species were defined as any species listed as occurring in the National Vegetation Classification communities (Rodwell 1991; Rodwell 1992) that were defined as floodplain grazing marsh by (Mountford et al. 2006) (Appendix D.2).

The presence of cattle dung in a quadrat was used to indicate the utility of the habitat for beef production, as dung is distributed in proportion to the time that cattle spend in an area (Jansen and Roberston 2001). Quadrats were searched for dung on two occasions, once in July 2012 and once in April 2013.

Three aspects of the aesthetic experience were quantified as ecosystem service indicators. Odonates and wetland birds can have positive impacts on recreational experiences (Lemelin 2009; Green and Elmberg 2014), while debris items can negatively affect recreational experiences (Williams and Simmons 1999). The quantity of debris present within each quadrat was counted on three occasions following substantial periods of flooding (March 2012, July 2012, May 2013). Debris commonly formed aggregations and so was recorded by surface area as the number of 0.3 m by 0.3 m grid squares that contained debris. The abundance of odonates was recorded following three minute search periods within each quadrat. Each quadrat was surveyed three times in suitable weather conditions in August 2012, and a further three times in August 2013. The activity of wetland bird species (*Anas clyptea, Anas crecca, Anas platyrhynchos, Anas strepera, Anser anser, Ardea cinerea, Aythya fuligula, Cygnus olor, Fulica atra, Haematopus ostralegus, Tadorna tadorna, Vanellus vanellus)* was recorded within each quadrat on six occasions in June 2012, and on a further six occasions between May and June 2013. The proportion of the twelve visits when wetland birds were recorded within a quadrat provides an index of habitat preference, but also a tangible recreational ecosystem service indicator, namely, the probability of a visitor observing a wetland bird during a visit.

The volume available for use as flood storage throughout the year is an indicator of the contribution that a floodplain patch makes to flood regulation. The topography of the site has been characterised in detail, and was combined with a series of water level measurements taken approximately every two weeks between 04/05/2011 and 19/04/2013 (Richards et al. in press). The volume available between the top of the flood defence banks and either the water level or topographic surface, depending on which was higher, was calculated for each quadrat on each occasion. The mean of these temporal measurements of flood storage volume was then taken as the flood storage indicator.

Environmental data were collected to model the provision of ecosystem service indicators. An index of flood exposure was mapped continuously across the floodplain as the number of hydrological survey dates that the area was underwater (minimum possible score =0, maximum possible score = 52) (Richards et al. in press). The mean flood exposure score was then calculated within each quadrat. A least-cost distance method was used to calculate an index of the flow distance between each quadrat and the closest flood entry point (defined as the bank breaches made during hydrological restoration) (Appendix D.1). Vegetation height within each quadrat was quantified as the mean of five measurements (one at each corner and the centre) which were taken in June 2012 using the method of (Stammel et al. 2003). The mean slope within each quadrat, and the mean altitude, were calculated directly from the topographic data.

6.3.3. Relationships between ecosystem service indicators and degree of flood exposure

Each ecosystem service indicator was modelled in response to flood exposure and additional explanatory variables that were expected to be relevant. FGM similarity and BAP species richness were modelled in response to flood exposure and cattle dung density. The presence of cattle dung was modelled in response to flood exposure and slope. The presence of odonates and wetland birds was modelled in response to flood exposure and vegetation height. The presence of debris items was modelled in response to flood exposure, vegetation height, and flow distance from the nearest flood entry point. Flood storage capacity was modelled in response to flood exposure and altitude.

Generalised linear models with binomial error structures were used to model the proportion of floodplain grazing marsh species that were present, and the proportional occurrence of cattle dung, odonates, litter items and wetland birds. A generalised linear model with a poisson error structure was used to model the number of plant BAP species that were present, and a linear model was used to model the volume of flood storage available. The presence of quadratic relationships between flood exposure and all the ecosystem service indicators was tested prior to building the maximal models. Quadratic terms for flood exposure were included in a maximal model if the quadratic model gave a lower AIC value than the simple linear model.

Spatial autocorrelation was present in all indicator datasets, so spatial eigenvector mapping was used to create spatial predictor variables that were then included in the models. We used the data-driven approach proposed by (Dray et al. 2006) to define truncated connectivity matrices based on Euclidean distance for each service indicator separately. Moran eigenvector filtering for the defined connectivity matrix was then applied to the maximal regression model for each service indicator to select a subset of spatial eigenvectors that removed significant autocorrelation from the model residuals. The selected eigenvectors were then added to the maximal model, and this model was simplified using a backwards stepwise procedure using An Information Criterion (AIC) as the simplification criterion (Dray et al. 2006). Map processing was conducted using the sp and raster packages for R (Pebesma and Bivand 2005; Bivand et al. 2007; Hijmans and van Etten 2012), and spatial eigenvector mapping was conducted using the spdep package (Bivand 2013).

6.3.4. Simulation process and analysis of optimality and heterogeneity

To investigate the optimality and multifunctionality of floodplain mosaics we simulated a range of hypothetical management scenarios. In each comparison of multiple ecosystem services, a range of hypothetical floodplain mosaics were simulated and then optimised, to identify the decision space and define the Pareto frontier.

Floodplain mosaics were simulated using a resample-with-replacement procedure. Groups of ten of the 100 survey quadrats were randomly sampled and combined to simulate a mosaic that was 0.02 km^2 in area. The value of five ecosystem service indicators was then quantified for the simulated mosaic. It was not possible to quantify flood storage or debris presence at the mosaic scale because these indicators are likely to be influenced largely by the surrounding topography and location, factors which were not modelled as part of the simulation. The five remaining ecosystem service indicators were quantified slightly differently for the mosaic-scale simulations than for the quadrat-scale statistical modelling discussed above. Similarity to FGM was again measured as the proportional occurrence of FGM species, and a species was counted as present in the mosaic if it was present at any quadrat ("patch") within it. BAP plant species richness was also measured over the whole mosaic. The suitability of each simulated mosaic for cattle production was measured as the proportion of the sample quadrats where cattle dung was present, as this provides an estimate of the available grazing area for cattle. The recreational benefit provided by odonates was quantified as the total abundance found in the resampled quadrats over all six survey dates. The probability of sighting a wetland bird anywhere on the simulated mosaic was calculated over the twelve survey dates.

The simulated floodplain mosaics were optimised for each pair of the five service ecosystem service indicators separately using an iterative process. For each pair of indicators a starting population of 1,000,000 floodplains was randomly generated, and the Pareto frontier for this starting population was calculated. The optimal floodplains from the original Pareto frontier were then mutated to form a second generation of 300,000 floodplains. For each mutation event, a random number of floodplain patches in the mosaic were replaced with patches that were sampled randomly from the pool of 100 survey quadrats. This process was repeated until the frontier stabilised, and after each iteration a subset of the floodplains in the decision space was recorded for further analysis. In addition to the ten pairwise optimisations, a similar optimisation was carried out for all five ecosystem service indicators together, the only difference being that this analysis used generation sizes of 1,000,000 floodplains. The optimisations were conducted in R, using an implementation of Pareto frontier calculation that was originally written in the Python language (Bull 2012).

For each optimisation, a range of floodplains were simulated within a decision space. The closeness to optimality of each simulated floodplain was quantified as the inverse of the minimum scaled Pythagorean distance from the Pareto frontier. The multifunctionality of the optimal floodplains on the Pareto frontier was quantified as the evenness (Pielou's J) (Zar 2010) of the suite of ecosystem services provided, with each service normalised as the proportion of the maximum value encountered on the Pareto frontier. The heterogeneity of floodplain mosaics was quantified as the number of different hydrological habitat types that were present (i.e. habitat richness). Habitat types were defined by categorising the gradient of flood exposure into 52 equal-sized groups.

The relationships between habitat heterogeneity and the relative optimality of floodplains were analysed as Spearman's rank correlation coefficient (ρ). The relationships between the habitat heterogeneity and the multifunctionality (J) of the optimal floodplains were also compared using Spearman's ρ. Only optimal scenarios were analysed in relation to multifunctionality, to control for variation that was due to optimality. It was not possible to assess the significance of the correlations between heterogeneity and optimality because the sample size, and therefore the number of degrees of freedom, was arbitrary. Similarly, the sample size when comparing heterogeneity and multifunctionality was often very small, as it depended on the number of simulated mosaics that were selected for the Pareto frontier. For both of these sets of analyses, ρ values are displayed without an estimate of statistical significance. We must be cautious about drawing general conclusions about the relationships between the studied ecosystem services, but within the context of the simulation we assume that the magnitude and direction of the correlations are correct. Confidence in the conclusions of the habitat heterogeneity vs. optimality correlations is increased because the simulated floodplains used in the analyses are a subset of the decision space that were sampled following an extensive, iterative procedure that searched millions of possibilities. The subset is therefore highly likely to represent the simulation system well. Confidence in the habitat heterogeneity vs. ecosystem service evenness correlations is also increased because although these analyses used small samples sizes, the floodplains used in the analyses were all optimal, thus minimising any confounding variation due to differences in optimality.

6.4. Results

All seven ecosystem service indicators responded significantly to the hydrological gradient (Table 6.1; indicative relationships between flood exposure and each service indicator are shown on Figure 6.2). Flood exposure was the most significant predictor of all service indicators (Table 6.1; for full details of the regression models see Appendix D.3). The proportion of floodplain grazing marsh plant species in the community, the number of BAP plant species present within a quadrat, and the probability of cattle dung and debris presence decreased significantly with increasing flood exposure (Table 6.1; also see Figures 6.2a, b, c, and e). The probability of sighting odonates and wetland birds showed significant unimodal responses to flood exposure (Table 6.1; Figures 6.2d, g), with peak values at flood exposures of 40 and 38 flooded occasions respectively (Figures 6.2d, g). Sites that experienced lower flood exposure also had significantly lower flood storage volumes available (Figure 6.2f).

When analysing the relationships between pairs of ecosystem service indicators at the scale of floodplain mosaics, Pareto frontiers were found in eight of the ten cases (Figures 6.3a:h) indicating that there was some level of trade-off between the provision of these pairs of services. Two pairs of ecosystem services were synergistic, as there were optimal floodplain mosaic configurations that allowed the provision of both services to be maximised together. These two pairs of ecosystem service indicators were FGM similarity and cattle suitability (Figure 6.3i) and wetland bird presence and BAP species richness (Figure 6.3j).

There was a negative correlation between hydrological heterogeneity and the closeness to optimality in nine of the ten pairwise comparisons (Table 6.2), although for the trade-off between FGM similarity and wetland bird probability this relationship was weak (Table 6.2). In the synergistic relationship between the probability of wetland bird presence and BAP species richness, scenarios that were closer to optimal were also more heterogeneous. The relationships between habitat heterogeneity and multifunctionality were compared for the simulated floodplain mosaics on the Pareto frontier, for the eight applicable pairs of services (i.e. excluding the two synergistic relationships). All pairs of services showed a positive correlation between habitat heterogeneity and multifunctionality (Table 6.2).

The five ecosystem service indicator optimisations showed similar patterns to the majority of the pairwise comparisons. Floodplains that were closer to optimality were also less heterogenous ($\rho = 0.43$). Amongst the six optimal floodplains, more heterogenous floodplains provided the five ecosystem service indicators more evenly ($\rho = 0.53$).

Table 6.1. Summary of regression models developed for each ecosystem service indicator at the quadrat scale. Explanatory variables that were retained in each final model following simplification by AIC are shown, with an indication of the statistical significance of their coefficient. Increasing numbers of asterisks indicates the statistical significance of the coeficient at the $p < 0.05$, $p < 0.01$, and $p<0.001$ levels respectively. For full details of the models including coefficient values and precise statistical significance, please see Appendix D.3.

Ecosystem service indicator	Explanatory variables in the final model
Floodplain grazing marsh similarity	Flood exposure**
Number of BAP species present	Flood exposure***
Cattle dung presence	Flood exposure***
Odonate presence	Flood exposure***+ flood exposure squared*** + vegetation height + spatial predictor***
Presence of debris	Flood exposure** + vegetation height + flow distance from nearest flood entry point
Flood storage volume	Flood exposure*** + flood exposure squared*** + spatial predictor***
Wetland bird presence	Flood exposure*** + flood exposure squared*** + vegetation height*

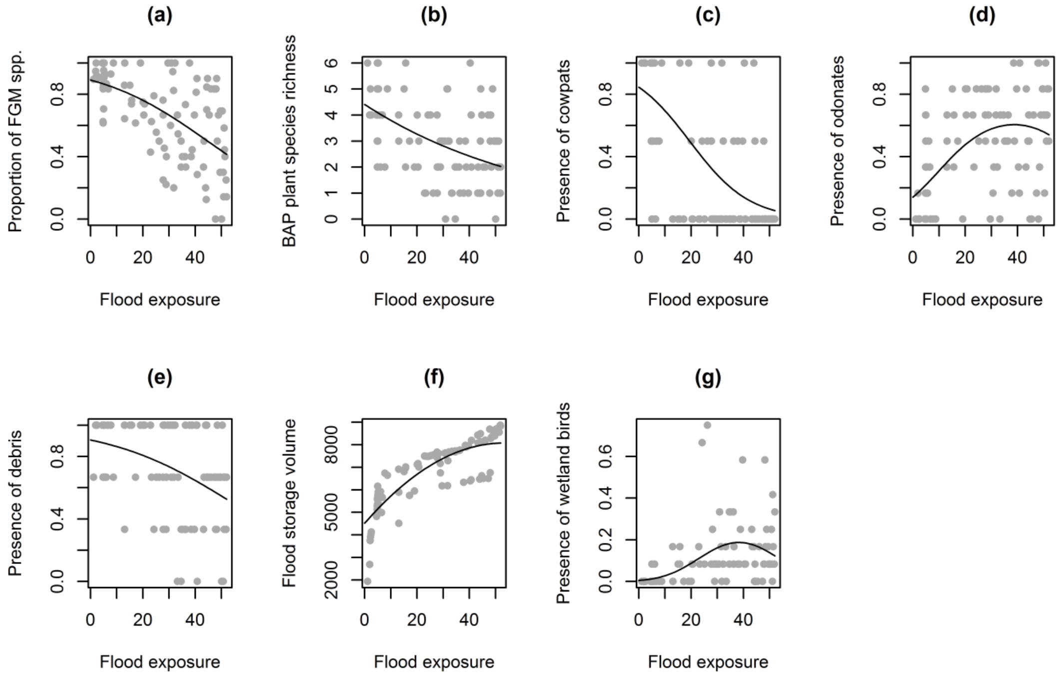

Figure 6.2. Indicative relationships between seven ecosystem service indicators and flood exposure (the mean number of 52 survey occasions when the areas within the quadrat was flooded). Black fitted lines in these figures correspond to the response variable modelled as a function of flood exposure only. Figures (d), (f), and (g) model flood exposure as a second degree polynomial.

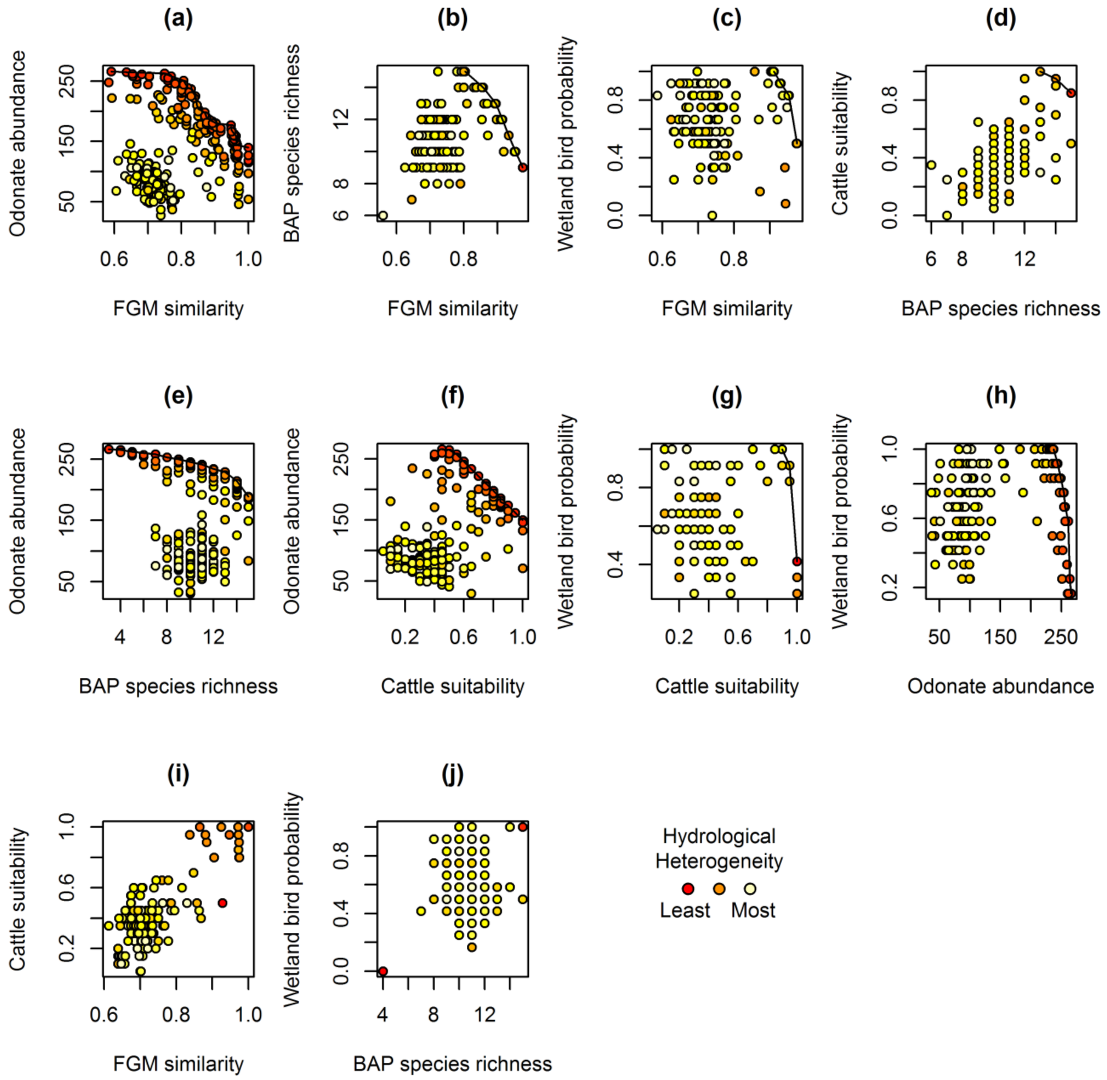

Figure 6.3. Pairwise trade-off or synergy relationships between the provision of five ecosystem service indicators in floodplain mosaics. Each point represents a simulated floodplain, coloured by the number of hydrological habitat types present within it. The ideal optimal solution in each case is in the top right corner. In each relationship Pareto frontiers are marked by black lines, if applicable. Heterogeneity is coloured on a white-red spectrum. More heterogeneous floodplain mosaics are coloured white, while less heterogeneous mosaics are red.

Table 6.2. Relationships between habitat heterogeneity and two general characteristics of ecosystem service provision; optimality and evenness. Figure numbers correspond to the trade-off plots shown in Figure 6.3.

Figure number	Service indicator 1	Service indicator 2	Spearman's ρ (heterogeneity against optimality)	Spearman's ρ (heterogeneity against evenness)
6.3a	FGM similarity	Odonate abundance	-0.85	0.39
6.3b	FGM similarity	BAP species richness	-0.33	0.95
6.3c	FGM similarity	Wetland bird probability	-0.11	0.99
6.3d	BAP species richness	Cattle suitability	-0.34	0.76
6.3e	BAP species richness	Odonate abundance	-0.7	0.68
6.3f	Cattle suitability	Odonate abundance	-0.79	0.67
6.3g	Cattle suitability	Wetland bird probability	-0.3	0.99
6.3h	Odonate abundance	Wetland bird probability	-0.52	0.55
6.3i	FGM similarity	Cattle suitability	-0.56	NA
6.3j	BAP species richness	Wetland bird probability	0.25	NA

6.5. Discussion

Hydrology has a strong influence on the ecosystem services that a patch of floodplain provides (Morris et al. 2009), and in this study the provision of services was separated into two bundles across the hydrological gradient. Drier habitat patches benefitted the two plant biodiversity indicators because they provided suitable habitat for greater numbers of wet grassland plant species (Rodwell 1991; Rodwell 1992 Wheeler et al. 2004), many of which were distinctive of floodplain grazing marshes, BAP-listed species, or both. Drier floodplain patches were also preferred by cattle, probably because grassland vegetation is more palatable (Buss et al. 2012), and locomotion in wetter environments is more difficult and hazardous (Ballard and Krueger 2005). On the other hand, wetter areas provided more suitable habitats for wetland birds and odonates (Buckton and Ormerod 1997; Pond Action 2000; Everard and Noble 2008), taxa which are often of recreational interest (Green and Elmberg 2014). Areas with higher flood exposure also accumulated fewer debris items, because debris tends to become stranded at high water marks, which are flooded infrequently (Storrier et al. 2007). The results for the flood storage indicator cannot be interpreted in the same way as the other service indicators. The volume available for flood storage was greatest in the quadrats that had the greatest flood exposure. However, this relationship is not causative, but rather a result of the fact that the most frequently wet areas of the site were located in the topographic low points. Mechanistically, the flood storage volume available in these areas would actually increase if flooding was less frequent because on average there would be less water filling the floodplain. The trade-off between low-intensity agriculture and grassland biodiversity in dry areas, and wetland biodiversity and recreational interest in wetter areas, is common in European floodplains (Rouquette et al. 2011; Sanon et al. 2012). Trade-offs between ecosystem services were driven mainly by the gradient of flood exposure, but direct interactions between ecosystem service indicators may strengthen these relationships (Bennett et al. 2009). For example, cattle grazing can affect plant community composition (Schaich et al. 2010c), and disturb birds (Paine et al. 1996).

The hydrological heterogeneity of the simulated floodplain mosaics affected the provision of ecosystem services because of the underlying relationships between flood exposure and service provision. Different quadrats varied in the contribution that they made to the ecosystem service provision of a mosaic, and some quadrats were superior to others in the context of each simulation. Some quadrats were therefore redundant; the service provision of a mosaic could be improved by replacing them. Highly heterogeneous floodplain mosaics tended to be suboptimal because, by chance, they contained greater numbers of redundant patches. The optimisation process gradually replaced the redundant patches to enhance ecosystem service provision, and because the provision of services depended on hydrology, optimisation tended to select groups of patches that had similar (or identical, as resampling was conducted with replacement) flood exposures. Optimal floodplains were thus hydrologically specialised to provide their target services, and as a result were comparatively homogenous. The negative relationship between habitat heterogeneity and ecosystem service optimality is likely to be general, because most ecosystem services require specific environmental conditions (Maskell et al. 2013), and are enhanced when there is a greater area of suitable habitat (Barbier et al. 2008; Smukler et al. 2010).

Optimal floodplain mosaics were less heterogeneous than many suboptimal ones, but there was variation in heterogeneity among the optimal mosaics. Floodplains that were optimised and monofunctional were the most homogenous, because they were composed exclusively of habitat patches that provided the environmental conditions necessary for one service indicator. On the other hand, optimal mosaics that provided an even balance of multiple ecosystem service indictors were more heterogenous. More heterogeneous habitats were more multifunctional because contrasting hydrological conditions provided different ecosystem service indicators (Mander et al. 2007). Multifunctionality may not always be enhanced by habitat heterogeneity, for example, in cases where multiple ecosystem services require similar habitat conditions (i.e. synergies). However, different ecosystem services commonly require different habitat conditions, resulting in trade-offs in provision (Rodríguez et al. 2006; Bennett et al. 2009), and the provision of many services scales positively with increasing habitat area (Barbier et al. 2008; Smukler et al. 2010). It is therefore likely that habitat heterogeneity will be required to provide multifunctionality (Mander et al. 2007).

In reality the majority of habitats are likely to be suboptimal in providing ecosystem services, because habitat management is subject to a range of physical, legal, and financial constraints (Rohde et al. 2006). The findings of the present study suggest that to improve the management of ecosystem services towards optimality, it is necessary to understand the specific environmental conditions that each service requires. It may be easier to provide optimal, monofunctional habitats, because the habitat conditions that are required to provide only one service must be understood, and it is not necessary to understand any trade-offs. Optimal, multifunctional habitats may be more complex to design because the habitat requirements, scaling relationships, and interactions between a number of ecosystem services must be known (Lovell and Johnston 2009; O'Farrell and Anderson 2010). While data on the physical and ecological drivers of ecosystem service provision can be lacking (Kremen 2005), in many cases such drivers are understood at least broadly (Wheaton et al. 2008), so it may be possible to approximate the effects of management with enough precision to create outcomes that are close to optimal, or at least satisfactory. Alternatively, adaptive management could be applied to fine-tune habitat management over time, to optimise the provision of multiple ecosystem services simultaneously (Kremen 2005; Folke et al. 2005).

Multifunctionality may be achieved in many cases through managing habitats to increase heterogeneity. However, an understanding of the habitat conditions that each ecosystem service requires is still necessary, as there are likely to be some habitat types that are redundant with respect to the target services. Furthermore, an understanding of the spatial scaling relationships of the individual ecosystem services is required so that decisions can be made about the proportional cover of different habitat types (Barbier et al. 2008). In reality, habitat management is unlikely to target maximum evenness in ecosystem service provision, but will attempt to balance the provision of different ecosystem services in a way that is satisfactory to the relevant stakeholders (Ananda and Herath 2009). Information on the weighting that stakeholders place on different ecosystem services would therefore be critical when precisely designing habitat heterogeneity. More generally, it is encouraging that some level of multifunctionality appears to be very common; even the scenarios that optimised one service provided some level of others (Figure 6.3). In some trade-offs, substantial levels of apparently conflicting service indicators could be provided together. For example, one optimal floodplain mosaic was entirely suitable for cattle production, and also provided suitable habitat for odonates (Figure 6.3f), despite the different hydrological requirements of these service indicators.

The simulation methodology used in this study did not take into account the spatial structure of the floodplain mosaics. Spatial structure is likely to have an impact on the ecosystem services provided, as some services are affected by habitat connectivity (Mitchell et al. 2013), or require different kinds of habitat to be adjacent to each other; for example, wetland birds require open water with adjacent vegetated banksides (Buckton and Ormerod 1997). It was not possible to model such aspects of spatial structure in this study, presenting opportunities for future research. Investigations of the impacts of other aspects of habitat structure, such as habitat connectivity, or beta diversity, could help to better understand the relationships between the general characteristics of habitat composition and the ecosystem services that they provide.

6.6. Conclusion

This study has shown that the hydrological heterogeneity of a floodplain mosaic has an impact on both the optimality of the ecosystem services it provides, and its multifunctionality. Some level of multifunctionality is likely to be almost ubiquitous in habitat mosaics, and notable levels of multiple service indicators can be provided even when they require very different environmental conditions. Habitat management that aims to provide multifunctionality should create heterogeneity in habitat conditions, but should ensure the provision of suitable habitat conditions for the specific ecosystem services that are desired. Creating habitat heterogeneity without an understanding of the specific requirements of the target services is risky, because there is a chance that the resulting suites of services will be far from optimal.

7. Impacts of ecosystem service information on people’s preferences for habitat management

The work of previous chapters improved our understanding of the relationships between floodplain management and ecosystem service provision. The present chapter investigates the ways that such information might be applied by decision makers, and thus assesses the impacts that ecosystem service frameworks may have on habitat management practices. It is important to understand how ecosystem service frameworks are likely to affect habitat management, because such frameworks are increasingly recommended in policy and guidance. This study used knowledge of four ecosystem services that were investigated in Chapters 4, 5, and 6, and combined an understanding of how these services work at Fishlake with knowledge gained from the wider literature. This information was used to hypothesise seven alternative management scenarios for use in a public decision making exercise.

This study compared the variability of groups of people’s preferences for habitat management scenarios. Different treatment groups were provided with contrasting detail on the ecosystem service impacts of the scenarios. People’s preferences for management scenarios are important in determining the decisions that are made, so variability in preferences may have implications for management practices. This study has been prepared for submission to Conservation Letters, and the manuscript has been lightly edited and extended for this thesis. The sections have been re-numbered and text has been formatted for consistency with the thesis, and all references can be found in the general references section.

7.1. Abstract

Detailed information on the trade-offs and synergies between the provision of ecosystem services should help to inform a more balanced approach to habitat management. However, the impact of ecosystem service information on habitat management decision making is not well known. When people have an understanding of the impacts of management on greater numbers of ecosystem services, there may be more variability between their preferences for habitat management options. People may show more variable preferences when provided with ecosystem service information firstly because such information can highlight the benefits of management options that would otherwise appear suboptimal. Second, if people are provided with information on a greater number of ecosystem services then they must balance a greater number of trade-offs between services. People balance trade-offs based on their biases towards the various ecosystem services that they are aware of, and with more biases to consider there is a greater chance that variability in biases between people will lead them to show different preferences. In this study the preferences that people showed for different floodplain management scenarios were analysed using an online participatory decision making exercise. Participants were provided with different amounts of ecosystem service information, and the variability of preferences within groups was analysed. The treatment group that was provided with the most detailed information exhibited the greatest variation in responses. More variable preferences for habitat management scenarios amongst a group of people could have implications for habitat management decision making and may lead to more multifunctional management practices.

7.2. Introduction

Historically, habitats have commonly been managed for specific purposes, such as the production of food or clean water (Raudsepp-Hearne et al. 2010b). This has led to degradation of many of the less obvious benefits that habitats can provide (i.e. ecosystem services) (Tallis and Polasky 2009). In contrast, an ecosystem service framework should inform people about a broader range of benefits, and thus reveal the trade-offs between different uses of a habitat (Raudsepp-Hearne et al. 2010a). An awareness of multiple trade-offs is expected to result in more holistic decision making, which may encourage multifunctional habitat management (Fish 2011). Ecosystem service frameworks are increasingly recommended in habitat management policy and guidance (Fisher et al. 2008; Fish and Haines-Young 2011). However, little is known about the impacts that such frameworks can have on management decision making; there are relatively few well-documented case studies (Goldman et al. 2008; Laurans et al. 2013; Ruckelshaus et al. in press), and fewer experimental tests (although see Shaw (2012)).

It is complex to predict how an ecosystem service framework will affect habitat management, because management decisions are made at a range of personal, organisational, and community scales (Bakus 1978; Fish and Haines-Young 2011) (Castles et al. 1971). People's preferences for different habitat management options can give an insight into the potential outcomes of decision making processes, because preferences will affect an individuals' actions and interactions with other people (Bakus 1982). The psychology of preferences is in itself complex (Simon 1952; March 1978), but we can assume that in general, people will prefer habitat management options that provide suites of ecosystem services that they consider to be important (March 1978). An individual's preferences for habitat management are based partly on their understanding of the different options, including their knowledge of the likely ecosystem service outcomes (Kørnøv and Thissen 2000). If people are acting rationally we may assume that they will avoid options that they believe to be suboptimal; options which can be improved upon without trading one service off against another (Tversky and Kahneman 1986). If there are multiple optimal scenarios, or if the optimality of the various management options is not clear, people will use the information that is available to them to decide how to balance trade-offs between multiple ecosystem services (Craik 1972; Koontz and Thomas 2006). This decision will be personal, because it will depend on relative value that an individual places on the various services (i.e. their bias toward different services) (March 1978; Hogan 2002).

Information on a greater number of ecosystem services may alter people's preferences for management options in two ways. First, options that appear suboptimal when few ecosystem services are considered may be revealed as being optimal when information on a broader range of services is available. Much ecosystem services research aims to make use of this mechanism; for example, studies that describe the non-market benefits provided by natural ecosystems hope that this information will increase interest in conservation (Armsworth et al. 2007; Laurans and Mermet in press). Second, additional ecosystem service information will alter people's perception of the net utility of a habitat management option, because people weight the value of each ecosystem service differently (Kumar and Kumar 2008). Increased information may therefore shift the balance of preferences between habitat management options (Costanza 2000; Laurans and Mermet in press).

Increasing the number of ecosystem services that are described is likely to alter individual preferences for management options, and this in turn may alter the pattern of preferences amongst a group of people. If a greater number of habitat management options appear optimal then fewer suboptimal options can be immediately discounted through logic, so people must consider the net value of a greater number of options. Additionally, people must weigh up the net balance of a wider range of ecosystem services, and a greater number of trade-offs, thus making the decision problem more complex (DeShazo and Fermo 2002). With a greater number of ecosystem services and trade-offs to consider there is more scope for an individuals' biases towards certain ecosystem services to lead them to make different decisions, and we might therefore expect to see increased variability in preferences amongst people.

Variability in preferences for habitat management options amongst a group of people may have direct consequences for habitat management. If the group of people are making a decision together then a range of opinions may help to form a more balanced consensus (Curtis and Lockwood 2000; van Wensem 2013). Alternatively, if the group of people are responsible for managing separate areas within a landscape then variable preferences may result in them choosing different strategies, thus creating spatial heterogeneity in the services that are provided (Hansson et al. 2005). This spatial heterogeneity in ecosystem service provision can allow multifunctionality at the landscape scale (Mander et al. 2007).

In this study we consider the impact that information about increasing numbers of ecosystem services can have on people's preferences for management options. We also investigate whether there is a difference in preferences when people are provided with quantitative data on the ecosystem service impacts of habitat management, compared to when they are made aware of additional ecosystem services qualitatively. We investigate these questions through a decision making exercise, based at a lowland river floodplain case study. Ecosystem service frameworks are of key interest for floodplain management (Morris et al. 2009; Posthumus et al. 2010; Sanon et al. 2012), as these environments have historically been intensively managed for a narrow range of purposes (Tockner and Stanford 2002), despite their potential to provide a greater range of services (Posthumus et al. 2010; Rouquette et al. 2011).

7.3. Methods

Individual preferences for habitat management options can be quantified in a number of ways; for example through informal discussion, structured or semi-structured interviews (Rowe and Frewer 2005), or fully factorial choice experiment surveys (Westerberg et al. 2010). The present study used a quantitative survey that asked participants to choose between pairs of competing management scenarios, in order to gauge the relative preference of each scenario. A quantitative approach was chosen so that the variability in preferences between participants could be analysed objectively. A quantitative choice experiment method was not used because this method would require participants to consider all possible combinations of ecosystem services, even if some combinations are not realistic because of the underlying physical or ecological mechanisms. This study instead asked participants to indicate their preferences for seven management scenarios that were developed based on an understanding of the effects of hydrology on service provision.

Students and staff from The University of Sheffield (United Kingdom) were recruited for an online decision making exercise over two periods; once in June and once in September 2013. Participants were asked to take on the role of a representative of a local community during a consultation on a floodplain management project, and were asked to make a series of pairwise choices between seven management scenarios. The scenarios were hypothetical but were based on knowledge of a real scheme: the Fishlake Wetlands project in South Yorkshire in the United Kingdom.

The provision of four ecosystem service indicators was considered in this study; the presence of European water voles (*Arvicola amphibius*), the capacity of the floodplain for regulating floods downstream, the species richness of wetland birds, and the herd size of beef cattle. The presence of water voles, a species of high conservation interest in the UK (Strachan et al. 2011), was used as an indicator of the biodiversity conservation value of the floodplain. The return frequency of a flood event that would cause minor damage to a downstream village was used as an indicator of the flood regulatory service provided by the floodplain (Wharton and Gilvear 2007). The size of the cattle herd that could be sustained on the floodplain was used as an indicator of the provision service of beef production (Morris and Brewin in press), and the species richness of wetland birds that were likely to be present was used as an indicator of wildlife recreation, because many wetland birds are of recreational interest (Green and Elmberg 2014).

Seven hypothetical floodplain management scenarios were developed, varying from a floodplain with almost continuous standing water (scenario A), to one that rarely flooded (scenario G). The impact of each scenario on ecosystem service provision was then estimated using knowledge of the relationships between flood frequency and ecosystem service provision. The seven scenarios were designed to be realistic, but also to fulfil the requirements of the experimental design, for example, none of the ecosystem services were completely synergistic with each other. This was to ensure that providing participants with information about increasing numbers of ecosystem services would make it necessary for them to consider a greater number of trade-offs.

Table 7.1. Factor levels for the seven floodplain management scenarios.

Scenario	Frequency of flooding in local village (years)	Water vole presence	Number of beef cows present	Wetland bird species richness
A	20	No	0	8
B	21	Yes	0	5
C	22	Yes	0	4
D	23	Yes	20	3
E	24	No	25	2
F	25	No	20	1
G	26	No	35	0

Water voles were predicted to be present at three of the four wetter scenarios (B, C, and D; Table 7.1) because the species requires standing water and associated bankside habitats (Strachan et al. 2011). Water voles were not predicted to be present at the wettest scenario because this was hypothesised as an extensive water body with high connectivity to the river and a dynamic shoreline, which would not provide the steep banks that water voles typically require (Strachan et al. 2011). Downstream flooding was expected to become linearly less frequent as the scenarios became drier (Table 7.1), because in drier scenarios it was expected that, on average, less of the floodplain would be filled with water. There would therefore be a greater capacity for water to be removed from the river and into the floodplain during periods of high flow. The frequency of flooding changed over a small range (every 20 - 26 years) because the regulatory capacity of most individual floodplain wetlands is relatively small; it is the net effect of multiple flood storage areas in a catchment that is of importance (Baek et al. 2012). A flood frequency in the range of 20 - 26 years is slightly more frequent than would normally be expected in the UK, but the Environment Agency's highest flood risk category includes flood return periods of 30 years (Environment Agency 2014). Additionally there was an experimental reason for using a relatively high flood return period; preliminary testing of factor levels found that participants tended to discount more realistic flood frequencies (e.g. 50 – 100 years; (Marsh 2008). Wetland bird species richness was assumed to be zero at the driest scenario, and to linearly increase up to the second wettest scenario (Table 7.1), as the diversity of wetland habitats was predicted in increase. The wettest scenario (A; Table 7.1) was hypothesised to provide habitat for considerably more wetland species due to the increased area of water bodies and provision of shoreline habitat suitable for wading birds (Rafe et al. 1985; Traut and Hostetler 2004). It was assumed that that it would not be economically feasible to graze beef at the floodplain below a certain threshold (20 cattle), and that beef production would thus only be possible at the driest four scenarios (D, E, F, and G; Table 7.1). The bimodal pattern of herd size over these scenarios (peaking at scenarios E and G; Table 7.1) is partially an experimental construct, as it was desirable that cattle production was not completely synergistic with flood regulation. It is feasible that the use of different breeds could give rise to this scenario; a more productive breed could be used in F and G, while a more hardy breed could be grazed in scenarios D and E.

Prior to beginning the decision making exercise, participants were introduced to the site and management problem and were provided with photographs and a brief text overview of a number of ecosystem service indicators, including a description of their importance to the local community. Participants were randomly assigned to one of three treatment groups of increasing information complexity, and were asked to make a series of choices between pairs of management scenarios, which were represented as text (see Appendix E1-3 for example questions). To reduce participant fatigue, each individual answered approximately half of the pairwise combinations in each treatment group (either 10 or 11 questions). A formal non-committal option was not included, but participants were able to ignore questions if they desired, providing an informal non-committal option (De Vaus 2002).

In the first treatment group (the simple treatment) participants were aware of a trade-off between flood risk and providing habitat for water voles (see Appendix E.1 for an example question). The driest management scenario (G) would provide the greatest capacity for removing flood water from the river at extreme levels, thus reducing flood frequency downstream at the local village most effectively (Table 7.1). On the other hand, the intermediately wet scenarios B, C, and D supported a water vole population, with D being the most balanced of these options because it was superior to scenarios B and C in terms of reducing flood frequency (Table 7.1).

In the second treatment group (intermediate treatment), participants were informed about a further two ecosystem services; the size of the cattle herd that could be sustained on the floodplain and the species richness of wetland birds that were likely to be present. These two ecosystem services were described in the introductory section of the survey, and participants were informed that some wetland birds require habitats similar to water voles, and that there is a conflict between management for water voles and for cattle. Participants therefore had an approximate understanding of the potential trade-offs between services, but were not informed of the detailed impacts of each management scenario on their provision (see Appendix E.2 for an example question).

In the third treatment group (complex treatment) participants were given detailed information describing the impacts of each management scenario on all four ecosystem services (see Appendix E.3 for an example question). According to this information, the management scenario that would best reduce flood risk (G) would also support the largest cattle population, but would provide no wetland bird species. Scenario D would provide habitat for water voles and would perform at an intermediate level for both providing wetland birds for recreation and supporting cattle (Table 7.1). Scenario A was designed to maximise wetland bird species richness, but this choice would not provide water voles and had the highest flood risk.In each treatment group the underlying floodplain management scenarios were therefore the same, but participants were provided with different amounts of information that described them.

Relative preference for each management scenario was measured as the proportion of times that the scenario was chosen, divided by the number of times that it was chosen plus the number of times that it "lost" against another scenario. Therefore, non-decisions in which the participant made no choice were excluded from analysis. To investigate whether the relative preference for scenarios differed between treatment groups, pairwise comparisons were made using Pearson's chi-squared contingency tables. In each case the expected frequencies were generated to take into account the relative response rates for different scenarios. To quantify variation in preferences within treatments, evenness of preference was calculated as Pielou's J of the proportional preference for each management scenario (Zar 2010). More variable preferences between participants would result in greater evenness, and higher J scores. The significance of differences in variability between treatments was assessed pairwise using a bootstrap method whereby the participant responses from the two treatments were pooled and resampled, and the difference in J between the two resampled populations was compared 1000 times (Edgington 1995).

To assess the potential bias of participants towards different ecosystem services, eight additional questions were asked during the survey. Participants were asked to indicate how positively they agreed with two statements about each ecosystem service, using a five-level Likert scale (statements are listed in Appendix E.4). Patterns in the correlation matrix of these responses were visualised using principal components analysis. To assess whether there were any systematic biases between the treatment groups, the Bray-Curtis similarities of participant bias responses were compared across the three treatment groups using ANOSIM. All statistical analyses were conducted in R (R Core Team 2012), and the vegan package was used to conduct ANOSIM (Oksanen et al 2007).

7.4. Results

In total the decision making exercise received 297 respondents who made 3054 preference choices, with responses split almost evenly between the two survey periods. The pattern of participant preferences did not noticeably differ between the two survey periods so they were pooled for further analyses. Response rate was approximately equal between the three treatment groups, but was greatest for the simple treatment (109 participants, 1134 decisions). The complex treatment received the second greatest response rate (99 participants, 1003 decisions) and the intermediate treatment received the lowest (89 participants, 917 decisions). The majority of participants (251 of 297) responded to the bias statements. Participants varied in their potential biases towards the four ecosystem services; principal component one grouped participants who were potentially biased towards water vole conservation and bird viewing (28% of variance explained, Figure 7.1a), while principal component two split these participants from those who were more likely to be biased towards flood regulation and beef production (22% of variance explained, Figure 7.1a). There was no significant difference in the responses of participants to the bias questions between the three treatment groups (ANOSIM; n = 251, R < 0.001, p = 0.463; Figure 7.1b).

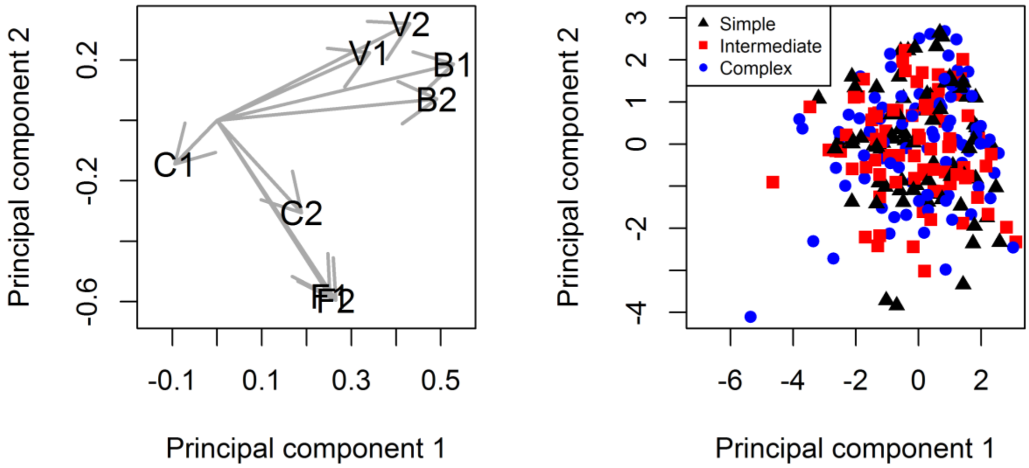

Figure 7.1. Principal components analysis of participant responses to bias questions. (a) Loadings of question variables for PC1 and PC2. V1 and V2 are loadings for questions about water vole bias, B1 and B2 correspond to questions about bird viewing, F1 and F2 are about flood risk bias, and C1 and C2 are about cattle farming. See SI4 for question details. (b) Participants plotted by their PC1 and PC2 scores.

In the simple and intermediate treatments, the strongest preferences were shown for the objectively optimal management scenarios; G, which had the least frequent flooding, and D, the scenario in which water voles were present that had least frequent flooding (Figure 7.2, top and centre panels; Table 7.1). The least preferred scenarios were A, which had the most frequent flooding and no water voles, and E, which also had no water voles and an intermediate flood return period (Figure 7.2, top and centre panels; Table 7.1). In the complex treatment the strongest preference was shown for management scenario B, which performed poorly for beef production and flood risk reduction, but relatively well for bird species richness, and supported water voles (Figure 7.2, bottom panel; Table 7.1). The marginally least preferred scenario in the complex treatment group was A, in which the local village flooded most frequently and there were no cattle or water voles present. However, scenario A provided habitat for the greatest number of wetland birds (Figure 7.2, bottom panel; Table 7.1). Relative preferences for the scenarios differed significantly between the simple and intermediate treatments ($X^2 = 492$, df = 13, $p < 0.001$), the intermediate and complex treatments ($X^2 = 1376.2$, df = 13, $p < 0.001$), and the simple and complex treatments ($X^2 = 508.9$, df = 13, $p < 0.001$). Variability in preferences between participants was lowest in the intermediate treatment group ($J = 0.933$), followed by the simple treatment group ($J = 0.943$). The complex treatment group showed the most variable preferences ($J = 0.987$). The variability in preferences of the simple and complex, and intermediate and complex treatment groups was significantly different following the bootstrap procedure at the $p < 0.001$ level. The variability of preferences in the simple and intermediate treatments was not significantly different ($p = 0.8$).

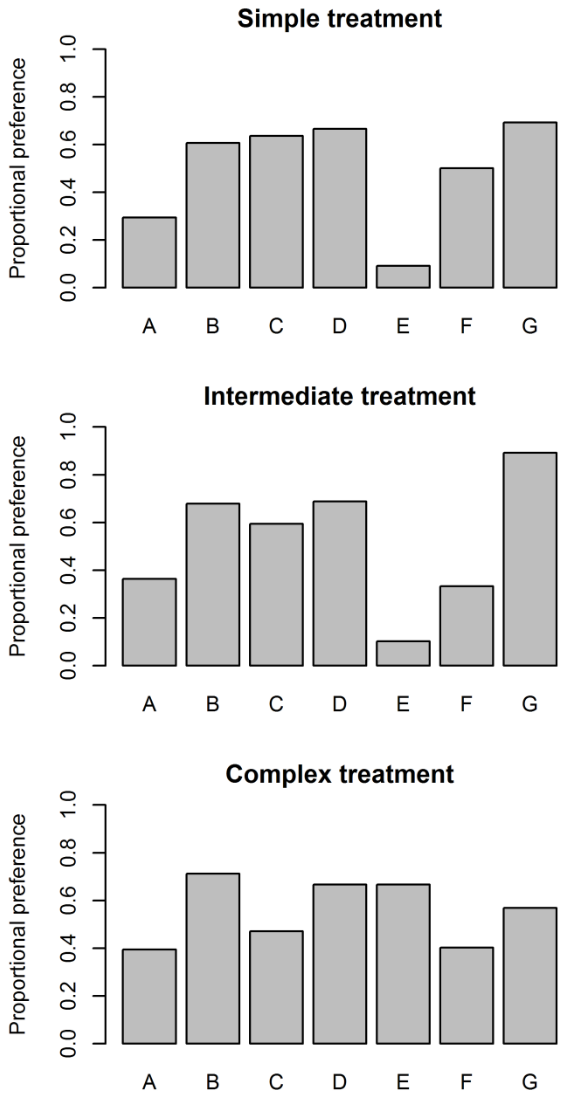

Figure 7.2. Proportional preferences for seven floodplain management scenarios in three treatment groups. The top panel indicates the simple treatment, the centre panel indicates the intermediate treatment group, and the bottom panel indicates the complex treatment group.

7.5. Discussion

Participants showed significantly different preferences for habitat management scenarios, depending on the ecosystem service information that was available to them. The group of participants that were provided with the most extensive ecosystem service information had the most variable preferences for management options. If these patterns in preferences for habitat management options hold true in different decision making contexts, they are likely to have implications for habitat management.

People's preferences for habitat management scenarios are affected by their understanding of the ecosystem service outcomes of the available options, and their personal biases towards particular ecosystem services (March 1978; Hogan 2002). In this study there was no significant difference in bias indicators between the treatment groups, so the observed differences in the pattern of preferences were most likely due to the different levels of ecosystem service information that participants were given. In the simple and intermediate treatment groups, participants preferred the optimal scenarios (G and D), indicating that they had utilised the available information to identify the trade-off between providing water vole habitat and reducing flood risk. Some choices favoured the scenarios that benefitted water voles, while others favoured flood risk reduction, indicating that personal biases led people to balance the trade-offs between these services in different ways.

It is interesting that participants in the simple and intermediate treatment groups showed no significant difference in variability of preferences for habitat management options, despite the additional ecosystem services that participants in the intermediate treatment group were aware of. It is possible that the coarse level of detail that was used to describe the additional services in the intermediate treatment may have led participants to consider them simply as either synergistic (wetland bird species richness) or conflicting (cattle herd size) with water vole presence. This broad understanding of ecosystem services would not reveal any further trade-offs between the services that could be provided, so would not greatly increase the complexity of the decisions that participants made (Ananda and Herath 2009). It has previously been suggested that qualitative descriptions of ecosystem service impacts may be a cost-effective way to implement ecosystem service frameworks (Busch et al. 2012), but the results of the present study suggest that a detailed knowledge of the ecosystem service impacts of management is required to alter preferences for management options (Naidoo et al. 2008; Aronson et al. 2010).

Participants in the complex treatment had the most variable preferences for habitat management scenarios. It is possible that participants had more variable preferences because the additional information confused them, leading them to select options at random (de Palma et al. 1994). However, this is unlikely because while the complex treatment involved more information than the simple or intermediate treatments, it presented a comparatively simple choice problem. A major study of information overload found no evidence of participant confusion with less than 10 options (i.e. scenarios), or 15 attributes (i.e. ecosystem services) (Malhotra 1982). The additional information in the complex treatment revealed the benefits of the management scenario that appeared suboptimal in the simple treatment (A); thus making it more attractive. Additionally, the broader range of information meant that people had to consider a larger number of biases when making decisions, and the decision was therefore more complex. Some participants were biased towards wetland bird species richness or cattle herd size, and consequently preferred management scenarios that prioritised one of these services; for example, scenario A gave the best outcome in terms of wetland bird species richness but performed poorly for the other services. Other participants balanced the complex series of trade-offs in a way that was desirable to them; for example, the most favoured scenario B balanced the trade-off between maximising wetland bird species richness and ensuring water vole presence.

The results of this study suggest that individual preferences for management options can be affected by an ecosystem service framework. However, it is not known how these individual preferences will affect decision making at the level of groups and organisations (Fish and Haines-Young 2011). The process of choice and decision making is well studied (Simon 1952; Bakus 1982; March 1978; Tonn et al. 2000), but there are few documented case studies relevant to habitat management (Koontz and Thomas 2006; Naidoo et al. 2009; Ruckelshaus et al. in press). The implications of variability in preferences for decision making will depend on the structure of the organisation making the decisions. In top-down management systems, for example when decisions are made by groups, variation in preferences among people may encourage consensus-building (Tonn et al. 2000; Fish and Haines-Young 2011; van Wensem 2013). The decisions that are made will depend on individual personalities and group composition (Gruenfeld et al. 1996; Kørnøv and Thissen 2000), but by making trade-offs more explicit and formalising the requirement to consider a wider range of benefits, an ecosystem service framework should support people who represent ecosystem services that have not been recognised in the past (Laurans and Mermet in press). In bottom-up management systems, where independent decision makers make a number of separate decisions, differing preferences for habitat management options may result in a mosaic of habitat practices, and thus ecosystem service provision, across a landscape (Hansson et al. 2005). The impacts of ecosystem service information on habitat management decision making will thus depend on the nature of the organisation making the decisions; top-down systems may be vulnerable to domination by powerful groups or individuals (Selman 1998; Holmes and Scoones 2001), while there is less opportunity for planning or consensus-building in bottom-up systems, so the results will depend largely on the preference and bias composition of the group, rather than being systematically organised to achieve specific targets.

It is hoped that ecosystem service frameworks will encourage habitat management that is based on a more informed understanding of the range of options that are available (Tallis and Polasky 2009; Fish and Haines-Young 2011). This study analysed the impacts of ecosystem service information on people's preferences for habitat management scenarios, but to further evaluate the utility of the approach, decision making at higher levels must be monitored and analysed (Koontz and Thomas 2006). Such research would better inform the application of ecosystem service information to real-world management problems.

8. General discussion

An ecosystem service framework should provide information that makes decision makers more aware of the broad range of benefits that habitats can provide, thus enabling them to predict the effects of habitat management on the provision of multiple services. This study aimed to inform the implementation of ecosystem services as a framework for managing habitats, with a particular focus on the hydrological management of floodplains. Specifically, it aimed to (1) provide a better understanding of the ecological impacts of increasing floodplain hydrological connectivity, (2) provide a framework for analysing the quality of recreational experiences at small spatial scales, (3) explore the effects of floodplain hydrological heterogeneity on ecosystem service optimality and habitat multifunctionality, and (4) investigate the effects of ecosystem service information at different levels of detail on people's preferences for floodplain management scenarios. This chapter reviews the principal findings of the study before discussing the implications for floodplain management and the implementation of ecosystem service frameworks more generally. To conclude, the chapter outlines future directions for implementation and research.

8.1 Principal findings

8.1.1. Effects of hydrology on floodplain ecology

Chapters 3 and 4 of this thesis analysed the relationships between hydrological conditions and two ecological components of floodplains. In Chapter 3, changes in plant community composition were analysed over a period of four years, during which the hydrological connectivity between the river and floodplain was increased. The plant community in the Fishlake floodplain changed in composition following reconnection, and the changes in composition could be explained by the functional traits of species. Changes in community composition were mainly due to changes in the abundance of species that were present in the floodplain prior to restoration, with limited colonisation by new species. Over the first, drier, two-year period of the study, competitive plant species increased in abundance and colonised new parts of the floodplain, while moisture-tolerant species generally declined in abundance. Over the second and wetter two year period, moisture tolerant species were more likely to increase in abundance while species with more competitive traits declined. Over both periods the relationships between functional traits and performance were not constant, but varied depending on the specific hydrological conditions at the sample plot. Overall, the plant community did not change significantly in relation to the target community of floodplain grazing marsh over the first two years of the study, but became significantly less similar to the target during the second two years of the study (between 2011-2013).

The findings of Chapter 3 are consistent with previous floodplain grassland studies in finding that colonisation by new species following hydrological restoration can be limited (Donath et al. 2003; Bissels et al. 2004; Rosenthal 2006; Gerard et al. 2008). It is likely that the trajectory of plant community change was constrained by the low rate of colonisation by new species, so future floodplain restoration projects must attempt to restore functional, ecological connectivity for source populations, as well as strucural, hydrological connectivity to the river.

In Chapter 4 the distribution and habitat preferences of European water voles (*Arvicola terrestris*) in the Fishlake floodplain was compared prior to and following the reconnection of the river (over a period of four years). Water voles preferred wider water bodies with taller and more diverse vegetation, and showed no range contractions within the active floodplain area, and an apparent expansion into one previously unoccupied area. Chapter 4 provides the first study of the habitat preferences of water voles in a floodplain wetland. Habitat preferences were consistent with the theory that extensive wetlands can provide suitable habitat for water voles, suggesting that restoration of wetlands could provide refuges for the species (Barreto et al. 1998a; Barreto et al. 1998b; Macdonald et al. 2002; Carter and Bright 2003). Previous reports have suggested that flood events can negatively impact water vole populations (Macpherson et al. 2003; Moorhouse et al. 2009), which could negate the positive effects of creating wetland habitats in floodplains. Chapter 4 provides a unique record of the impacts of floodplain reconnection on a resident water vole population, and reports no noticeable negative impact of substantial flooding on the water vole distribution in the floodplain. This indicates that water voles may be able to cope with the disturbance caused by flooding, and may thus benefit from future floodplain restoration projects (Barreto et al. 1998a).

8.1.2. Analysing the provision of recreational experiences at a fine spatial resolution

In Chapter 5 the provision and delivery of recreational experiences was investigated. In particular, this chapter focused on the importance of visitor interactions with habitat components in determining the quality of a recreational experience. The quality of the visitor experience was quantified in relation to one positive habitat component (odonates) and one negative habitat component (debris). Field data describing the spatial distribution and noticeability of the two habitat components were combined to estimate the number of each that a visitor standing at a particular location in the floodplain would be likely to experience. The net recreational impact of the habitat components was quantified using a willingness to pay approach which allowed the positive perception that people had of odonates to be traded-off against the negative perception that they had of debris items. The recreational experience model was used to demonstrate the importance of considering the interaction between habitat components and the visitor in determining experience quality. Models of recreational experience that did not consider the observation rate of odonates and debris gave substantially different results to those that considered this interaction between the visitor and the habitat. The development of the recreational experience model gave a better understanding of how recreational experiences are delivered to people, and the resulting experience model was used to predict recreational experience quality at a fine spatial resolution.

This study is among the first to consider the noticeability of habitat components when analysing the delivery of recreational experiences from habitats to people. The study also utilised techniques from environmental economics in a novel way to calculate an index of net recreational quality. The framework provided in this study could be a useful tool for designing habitat management, such as the design of wildlife viewing site locations, which was presented as a hypothetical example. The modelling framework could be expanded to consider more than two habitat components and applied to model recreational experience quality in other habitats.

8.1.3. The impact of hydrological heterogeneity on the provision of multiple ecosystem services

There is increasing interest in understanding how habitat management can affect the provision of multiple ecosystem services. In addition to understanding the specific trade-offs between particular ecosystem services, it can be useful to describe the characteristics of suites of services, for example the optimality of provision (Lautenbach et al. 2010) or multifunctionality of a habitat (Wiggering et al. 2006). Chapter 6 analysed the responses of seven floodplain ecosystem service indicators to a hydrological gradient; the similarity of the plant community to the target community (floodplain grazing marsh), the number plant species that were of Local Authority biodiversity conservation interest, the capacity for storing flood water from the river, and the probabilities of wetland bird presence, odonate (dragonflies and damselflies) presence, debris presence, and use by beef cattle. This chapter found that although each service responded differently to hydrological conditions, there were two broad trade-offs between beef production and grassland biodiversity in dry areas, and wetland biodiversity and recreational interest in wetter areas.

Chapter 6 synthesised information on five of the seven ecosystem service indicators (excluding flood storage volume and debris) to provide a better understanding of the responses of a whole suite of services to hydrology. The optimality of ecosystem service provision and the multifunctionality of simulated floodplain mosaics were analysed with respect to their hydrological heterogeneity. Hydrological hetereogeneity is a key feature of natural floodplains which is important in maintaining the biodiversity of these habitats (Ward et al. 1999). To investigate the relationships between heterogeneity and general characteristics of the suite of services provided, heterogeneity was correlated against the optimality of the ecosystem service provision, and the multifunctionality of the floodplain. In general, the habitats that provided ecosystem services optimally were less heterogeneous than those that were further from optimal, suggesting that some specialisation of floodplain mosaics is required to reach optimal provision. Amongst the optimal floodplain mosaics, those that were more heterogeneous were also more multifunctional, suggesting that diversity in habitat types can allow different ecosystem services to be provided by different parts of a habitat mosaic. Management of habitat heterogeneity for the purpose of multifunctionality may be a particularly useful strategy in floodplains, as natural floodplain habitats are characteristically heterogeneous (Ward et al. 1999) and that can provide a range of ecosystem services (Posthumus et al. 2010).

The grouping of floodplain ecosystem services into two main clusters is consistent with previous studies of trade-offs in floodplain ecosystem services (Rouquette et al. 2011; Sanon et al. 2012), and Chapter 6 additionally identified more subtle trade-offs between pairs of services. The chapter provides one of the first quantitative analyses of how general patterns in habitat composition, in this case habitat heterogeneity, can affect the broad characteristics of the suite of services a habitat provides. Optimal service provision or habitat multifunctionality are attributes that an ecosystem service approach might be expected to encourage (Chapter 1). This study provides information that may lead floodplains and other habitat types to be better managed to achieve optimality or multifunctionality.

8.1.4. Responses of decision makers to ecosystem service information

Chapter 7 took the opportunity provided by the Fishlake case study, and the research presented in the earlier chapters, to explore how ecosystem service information may affect habitat management decision making. Participants in a decision making exercise were asked to indicate their preferences for seven habitat management scenarios, based on different amounts of ecosystem service information. Participants in the simplest treatment group were provided with quantitative data about the impacts of management scenarios of two ecosystem services. The intermediate treatment group were informed about two additional ecosystem services, but were not given any quantitative information about these services. Participants in the most detailed treatment group were given quantitative information about the effects of management scenarios on all four ecosystem services. The information that people were provided with had a significant effect on their preferences for different habitat management options. An increased amount of information resulted in more variable preferences for habitat management scenarios, but only when the participants were provided with detailed, quantitative estimates of the effects of habitat management.

Little is known about the way that providing information about ecosystem services may affect habitat management decision making (Goldman et al. 2008; Laurans et al. 2013; Ruckelshaus et al. in press). This study showed that people responded to ecosystem service information and that their preferences for management scenarios were accordingly different. It is likely that people’s preferences for management options were more variable with information on more ecosystem services because individuals had personal ecosystem service priorities, so chose to balance trade-offs in ways that were desirable to them. More variable habitat management preferences within a group of decision makers could drive habitat management towards multifunctional outcomes or collaborative decision making (Tonn et al. 2000; Fish and Haines-Young 2011). However, decision making is a complex process that depends on the interaction between individuals, organisational structure, and socio-economic constraints (Bakus 1982; Fish and Haines-Young 2011), so it is difficult to predict the impacts of variability in preferences on a real-world habitat management.

8.2. Synthesis

The results from this study develop our understanding of some of the issues with implementing ecosystem service frameworks for managing habitats at local scales. The findings have implications for the management of lowland river floodplains in the UK and elsewhere, and in particular can inform the restoration of floodplains through hydrological reconnection. The study also provides insights into the mainstream implementation of ecosystem services frameworks more generally. As an intensive study of a specific site, this thesis provides a framework for collecting ecosystem service data, identifying trade-offs between services, and exploring the impacts of different management options on multiple services. The study as a whole raises technical questions about how an ecosystem service framework should be implemented, and provides insights into the ways that ecosystem service frameworks may affect habitat management practices.

8.2.1. Implications for the management of floodplains

Floodplains in the UK and Europe have been heavily modified by humans (Tockner and Stanford 2002, Maltby et al. 2011), but the past management paradigm of floodplain disconnection and agricultural intensification is now being replaced by a more multifunctional approach to management (Morris et al. 2004; Maltby 2009a; Morris et al. 2009). There is increasing interest in restoring more natural hydrological regimes in floodplains (Buijse et al. 2002; Tockner and Stanford 2002), and it is expected that hydrological restoration will provide suitable conditions for floodplain biodiversity (Schiemer et al. 1999). However, there is uncertainty about the impacts of floodplain restoration on biodiversity and ecosystem service provision (Maltby et al. 2011). Chapter 3 reports that over four years following the restoration of hydrological connectivity at Fishlake, the plant communities became less similar to the target of floodplain grazing marsh, and Chapter 4 suggests that there has been little change in the distribution of water voles in the floodplain over 3 years. However, ecological responses to restoration may take several decades (Donath et al. 2003; Bissels et al. 2004), and flooding at Fishlake was restored within only four years. Furthermore, the Fishlake floodplain was not restored from a completely degraded “blank canvas”; the site has historically received infrequent inundation, and there have always been some permanent water bodies present (Chapter 2). The quality of the ecological baseline may therefore have been high compared to restoration projects that re-wet arable land (Comín et al. 2001), so the changes brought about by restoration may be less extreme. Other examples of floodplain restoration are encouraging, as projects in Denmark (Lauge et al. 2007), Luxembourg (Schaich et al. 2010b; Schaich et al. 2010c) and Austria (Funk et al. 2009) have shown significant ecological changes towards their targets, over periods of between one and three years. The restoration of floodplains through hydrological reconnection thus has some potential to enhance wetland biodiversity (Zsuffa and Bogardi 1995; Fischenich and Morrow 2000; Buijse et al. 2002; Tockner and Stanford 2002; Toogood et al. 2008).

The Fishlake floodplain in its current state provides substantial biodiversity value, as it provides habitat for water voles (Chapter 4), wetland birds (Chapter 6), aquatic invertebrates (Chapter 5), and a diverse plant community (Chapter 4). These ecological resources provide opportunities for recreation (Chapter 5). Furthermore, the floodplain has maintained a flood storage capacity (Chapter 2) and supports the same size herd of cattle (Chapter 2, Chapter 6) as it did prior to restoration. Chapter 6 identified a general trade-off between floodplain patches that provide habitat for wetland birds and odonates, and floodplains that are more suitable for cattle production, provide a larger flood storage capacity, and provide suitable habitat for target plant species. Similar trade-off patterns are well known in floodplains (Rouquette et al. 2011; Sanon et al. 2012), but the Fishlake floodplain balances this trade-off to provide some level of multifunctionality.

The Fishlake floodplain is multifunctional at a local scale, but the ecosystem services that it provides also add diversity to the suite provided across the wider landscape. Floodplain wetlands can provide habitat refuges for wetland and aquatic species (Chapter 3, Chapter 4), and ecologically restored floodplains can provide resources for nature recreation (Chapter 5, Chapter 7) which may be rare in agricultural and urbanised landscapes (Sanon et al. 2012). These functions can be restored to highly modified floodplain landscapes by changing the management regime of a relatively small area, for example, Fishlake covers only a small part of the historical wetland area of the Humberhead Levels (Chapter 2). Hydrological restoration of small areas of floodplain could therefore restore some provision of biodiversity and cultural ecosystem services to a landscape, with only minor losses in agricultural capacity. This is an example at a larger spatial scale of the positive relationship between habitat heterogeneity and multifunctionality observed in Chapter 6.

The restoration of floodplains can add multifunctionality to landscapes, but the landscape context of a floodplain restoration project is also critical in determining the ecosystem services that it provides. Connectivity between floodplain patches is a key feature of natural river systems (Wiens 2002), and is important in determining the composition of ecological communities (Ward et al. 1999; Tockner et al. 1999; Reckendorfer et al. 2006). Without functional connectivity to similar habitats, the ecological restoration of floodplains can be limited (Chapter 3), and this may constrain changes in ecosystem service provision. The configuration of habitats in a landscape can also affect recreation, because recreational quality can be higher if habitat patches are connected to form longer walking or cycling routes (Briffett 2001). The position of a floodplain in its catchment can affect its flood regulatory capacity, because a network of interconnected flood storage areas is required to effectively manage river levels (Baek et al. 2012). In the past, habitat restoration has often been conducted opportunistically as funding or land opportunities arise (Holmes and Nielsen 1998; Rohde et al. 2006), but this strategy is unlikely to provide the required degree of connectivity between floodplain patches. Ideally, future floodplain restoration would be planned with consideration of the state of the rest of the catchment and surrounding landscape (Harper et al. 1999; Wohl et al. 2005; Rohde et al. 2006), and river management is increasingly moving in this direction (Mainstone and Holmes 2010; Gilvear et al. 2013). At the very least, the management of a single floodplain site should consider the benefits that habitat management could bring to the wider landscape, and any constraints that the position in the catchment may impose.

8.2.2. Implications for the application of ecosystem services as a framework for habitat management

This study has analysed ecosystem service provision at a local scale, across a gradient of habitat management. This study helps to bridge the gap between broad analyses of ecosystem service provision at large spatial scales (Costanza et al. 1997; Chan et al. 2006; Raudsepp-Hearne et al. 2010), and highly detailed analyses of only one or two ecosystem services (Brönmark and Hansson 2002; Grossmann 2012), to inform the management of multiple services at a single site. The functional assessment approach (Maltby 2009b) provided a framework for the rapid initial evaluation of wetland ecosystem services, which could be viewed as comparable in scope to the Phase 1 habitat surveys used in ecological monitoring under current legislation (Byron et al. 2011). Together the chapters of this thesis provide an example methodology for measuring and analysing ecosystem services at a local scale in considerably more detail, which may be more comparable to the existing Environmental or Ecological Impact Assessments (Byron et al. 2011, CIEEM 2013).

The chapters of this thesis focused on specific questions, but taken as a whole the separate studies provide an example of how data on ecosystem services could be collected to inform a habitat management problem, such as the design of a habitat restoration project. If ecosystem service frameworks become mainstream tools for habitat management at local scales, they will likely be implemented by existing environmental management organisations, such as wildlife charities, government bodies, and ecological consultancies (Byron et al. 2011). Such organisations are limited in resources (Wilson and Hoehn 2006), so ideally ecosystem service data should be straightforward and cost-effective to collect. This study focused on physical and ecological indicators of ecosystem services which are relatively simple to measure and interpret (Hernández-Morcillo et al. 2013). The majority of data collection did not require any specialist equipment or expertise (other than ecological field skills), so should be within the capabilities of a consultancy company (Byron et al. 2006; CIEEM 2013). This study has utilised large amounts of field data, but the data collection load could be reduced for site management purposes by reducing the number of temporal or spatial replicates. Additionally, some ecosystem service indicators may already be monitored under current guidance or legislation. For example, water voles and birds were surveyed during the Fishlake project scoping study, and the likely impacts of restoration on cattle farming and human safety were considered (Hiley et al. 2008; Natural England 2011). It might be that organisations can collect additional ecosystem service data at little cost during other surveys, for example by recording sightings of debris items, attractive organisms (Chapter 5), cattle dung (Chapter 6), or visitor numbers (Chapter 2). There is a risk that constraints on data collection may limit the breadth of ecosystem services that a project considers, but a combination of field data, spatial modelling (Nelson et al. 2009; Tallis and Polasky 2009; Guerry et al. 2012), and benefit transfer from the related literature (Dubgaard 2003; Wilson and Hoehn 2006) should provide information on a subset of services that is useful in informing habitat management.

Future ecosystem service frameworks for habitat management must collect information on multiple ecosystem services. However, it is equally important to consider the choice of index that is used to represent each ecosystem service. In all chapters of this thesis the choice of indicator was made with careful consideration for practicality and utility, but was subjective (Boyd and Banzhaf 2007). In Chapter 2, similarity to floodplain grazing marsh was chosen as the index of plant biodiversity "quality", and in Chapter 4, water vole presence was used as an indicator of the biodiversity value provided by this species. In Chapter 5, odonates and debris items were chosen to analyse recreational value, and a decision was made about the way that these two components were combined to measure net recreational experience quality. The choice of index is likely to have consequences for decision making; in Chapter 5 the results of the wildlife viewing site study were different depending on the index of recreational quality that was used. In Chapter 6 the way that ecosystem service indicators were measured, and the way that they scaled up from the patch to the mosaic scale, affected the trade-off relationships between different services. In Chapter 7, people's preferences for habitat management scenarios were different depending on whether they were provided with qualitative or quantitative descriptions of ecosystem service value.

An ecosystem service can be described using many different indicators (Hernández-Morcillo et al. 2013), but the findings of this study can be used to suggest three criteria for selecting useful ecosystem service indicators. First, ecosystem service indicators should ideally be quantitative, as qualitative indicators appear to have less impact on people's preferences for management scenarios (Chapter 7). Second, ecosystem service indicators should be measured in units that are easily interpretable in relation to the benefits that people gain. In Chapter 7, the species richness of wetland birds was described to participants in the decision making study. While expert wildlife enthusiasts may be more interested in the identities of the species that are present (Kerley et al. 2003), species richness is an index that does not require additional knowledge, so was appropriate for the non-specialist audience of the study. Third, ecosystem service indicators should, where possible, measure the net effect of habitats on the provision of services. In Chapter 5 the net recreational experience provided by two habitat components was quantified as a net debris balance. This indicator at first appears difficult to interpret, because people do not commonly think about their recreational experiences as a net balance. However, the concept of a balance is familiar to the public, and net metrics are advantageous because they allow the impacts of large numbers of habitat components to be summarised succinctly.

To apply an ecosystem service framework to habitat management decision making it is necessary to predict the impacts that different actions will have on service provision (Nelson et al. 2009). This study shows that environmental factors (Chapters 3, 4, and 6) and human perception (Chapter 5) can be used to predict the provision of ecosystem service indicators. However, there is still considerable uncertainty when predicting the responses of ecosystem service indicators to management (Failing et al. 2012). Uncertainty in the impacts of management is a problem for habitat management programmes that have specific ecosystem service targets (Wheaton et al. 2008; Palmer and Filoso 2009). In particular, uncertainty in the outcomes of habitat management may inadvertently lead to misleading management suggestions (Nicholson et al. 2009). Uncertainty in management outcomes may also affect the decisions that people make about habitat management. The uncertainty of habitat management outcomes will be considered during decision making, and the responses of some services will be more predictable than others (Ascough et al. 2008). Ecosystem services that are more unpredictable may be perceived as more "risky", which may reduce the likelihood that habitat management will be designed to target them (Maguire and Albright 2005). To balance the trade-off between the desirability of an ecosystem service and the uncertainty of a habitat management outcome, decision makers must know how predictable an outcome is (Wheaton et al. 2008). However, uncertainty is rarely characterised in habitat management, for example only 33% of river restoration practitioners claim to quantify uncertainty (Wheaton et al. 2006). One of the respondents in the decision making study (Chapter 7) did not consider the risk of flooding when choosing between management scenarios because they argued that "being experienced in flood risk, I would say that the confidence in being able to predict flood extents … is low". This respondent was able to use their own knowledge of uncertainty when making their decision, but many of the participants were not so experienced. Decision makers using an ecosystem service framework would ideally be provided with information on the uncertainty associated with each management scenario (Nicholson et al. 2009), but it can be complex to characterise uncertainty (Ascough et al. 2008). If the specific impacts of habitat management are unpredictable then it may be useful to also describe some general patterns that may be more certain. For example, if it is a general trend that habitat heterogeneity results in multifunctionality (Chapter 6), habitat managers may be fairly confident that they will be able to achieve this high-level target, which may give them more flexibility in making more specific predictions.

8.3. Future research directions for the application of ecosystem services

Habitat management, and in particular the ecological restoration of habitats, is likely to receive more attention in research, policy, and practice in the future (Buijse et al. 2002; Tockner and Stanford 2002). This study contributes to the understanding of ecosystem services as a framework for managing habitats at a local scale, and sets a start point for future research in two main areas. First, simulation approaches have potential as tools for predicting the impacts of habitat management, and exploring the range of possible management scenarios. Second, more research must investigate the impacts of ecosystem service frameworks on decision making, to predict trends in future habitat management.

In the future, ecosystem service frameworks must integrate understanding of the physical, ecological, and human aspects of service provision. They must also combine data to describe the provision of multiple ecosystem services in detail, and must allow decision makers to explore the range of management possibilities (Nelson et al. 2009). Simulation approaches such as those described in Chapters 5 and 6 could provide useful tools for predicting ecosystem service provision to inform management (D' Aquino et al. 2003; Le Bars and Le Grusse 2008). However, there are a number of improvements that could be made to the simulation methods used in these chapters. In particular it would be valuable to consider the spatial configuration of habitat patches, because factors such as connectivity are likely to affect the delivery of ecosystem services (Mitchell et al. 2013). Similarly, future simulations should attempt to place sites in the wider landscape, and predict the cumulative impact of multiple interventions (Rohde et al. 2006). Additionally, ecosystem service simulations should attempt to quantify the uncertainty in the estimates that they make (Ascough et al. 2008), for example through the use of bootstrap methods (Jiang et al. 2013).

Simulation of habitat management scenarios can be a useful tool for comparing the effects of specific actions, but there can be a bewilderingly large number of options to compare (Shaw 2012). The optimisation approach taken in Chapter 6 can be used to explore the decision space objectively, to identify trade-offs between services

(Lautenbach et al. 2010; Seppelt and Lautenbach 2010), and to identify patterns in habitat management that are likely to result in management that is closer to optimal, more multifunctional, or meets both of these criteria (Chapter 6). Pareto optimisation could become a useful tool for analysing ecosystem service data (Lautenbach et al. 2010; Seppelt and Lautenbach 2010), because the methods are highly flexible, and can be used with almost any kind of model that links habitat to ecosystem service provision. Qualitative (ordinal), statistical, mechanistic, resampling, and even coupled environment and human interaction models (Chapter 5) could be optimised.

If the concept of ecosystem services becomes the leading paradigm for habitat management, it will be important to understand the impacts that it may have on the way that habitats are managed. Therefore there is a need to better understand habitat management decision making processes (Chapter 7), and in particular to understand the impacts of ecosystem service information in such processes. A better understanding of decision making may allow ecosystem service information to be provided in the most useful format for habitat management, and may allow governments to design policy instruments that give predictable outcomes (Smajgl et al. 2011). Such knowledge is critical to ensure that ecosystem service frameworks do not have unexpected effects on the management of habitats in the long term.

8.4. Conclusions

This study provides knowledge that can inform the hydrological restoration of floodplains, and improve the implementation of ecosystem services frameworks for local habitat management. The main findings of the study are: (1) Floodplain plant community composition changed dynamically at the Fishlake floodplain over four years in response to hydrological conditions, but the trajectory of change was constrained by the availability of colonising species. (2) Water voles in a floodplain wetland preferred wide water bodies and tall, diverse vegetation. There was no evidence in the Fishlake case study that substantial flood events negatively affected the resident water vole population. (3) The interaction between people and habitat components such as debris items and odonates is important in determining the quality of their recreational experiences. This interaction can be modelled using data on the distribution of habitat components, the observation rate of habitat components, and the relative preferences of people for these components. (4) Floodplain mosaics that provided optimal suites of ecosystem services tended to be relatively homogenous. Amongst the optimal floodplain mosaics, more multifunctional mosaics were more heterogeneous. (5) People's preferences for different habitat management scenarios were different depending on the ecosystem service information that they were provided with. The treatment group that was provided with more detailed ecosystem service information showed more variable preferences for management scenarios.

Together, the chapters of this thesis provide information on the potential for the hydrological restoration of floodplains, as well as some of the constraints that must be considered. They show that there are inevitably trade-offs between the provision of different ecosystem services, but that floodplains have the potential to be managed multifunctionally. Restoration of hydrological connectivity in floodplains could increase habitat heterogeneity, and enhance the multifunctionality of the habitat; both locally and at the landscape scale. However, the results also show that it is important to consider the landscape context when designing and implementing floodplain habitat management.

The thesis documents a case study in applying ecosystem service methodologies to habitat management, and in particular can inform the data collection of future management projects. To conclude this final chapter I outline a seven-stage guidline for applying an ecosystem service approach to the restoration of floodplains at local scales:

1. Floodplain management decision makers must first identify the range of ecosystem services that are most likely to be relevant for the floodplain of interest. Comprehensive lists of ecosystem services that are commonly relevant in floodplain and wetland systems can be found in the Millenium Ecosystem Service Assessment (MA 2005), the UK National Ecosystem Assessment (Maltby et al. 2011), and the functional approach procedures outlined in Maltby (2009). These comprehensive lists may be simplified to a more site-specific list depending on the location, history, and spatial scale of the study floodplain. Lists of ecosystem services are more likely to fit the concerns of the local population if they are designed after consultation with local stakeholder groups.

2. Decision makers must identify the most relevant habitat resolution at which to analyse the provision of ecosystem services. In the case of local floodplain restoration projects a relatively fine level of habitat detail is required, and hydrology is likely to be the most relevant factor with which to identify habitat types. Habitat types could be separated into qualitatively defined hydrogeomorphic units (Maltby 2009b), or quantitatively defined using topographic data (Besnard et al. 2013) or field measurements of flood exposure (as applied in this study). It is important to consider the habitat resolution at an early stage so that the study best represents the habitat management interventions that are likely to be put in place, and enables ecosystem service indicators to be selected that are most relevant for the decision makers.

3. Decision makers must choose indicators that suitably represent each ecosystem service at the chosen habitat resolution. Indicators should be selected following the three criteria in section 8.2.2; they should be quantitative rather than qualitative, should be measured in units that can be readily interpreted by decision makers, and should summarise the net effect of multiple ecosystem processes or components on the value delivered to people. It is unlikely to be feasible to analyse of all the relevant

ecosystem services in a high level of detail, so some services must be prioritised. Services could be prioritised in relation to their importance to stakeholders and decision makers, with the highest-priority services being measured directly in the field. Ecosystem services that do not require such detailed quantification could be assessed using modelling tools such as InVEST (InVEST 2014) or EcoServ (Durham Wildlife Trust 2014), or estimated using benefit transfer techniques (Wilson and Hoehn 2006).

4. The collection, analysis, and modelling of ecosystem service data is likely to be the most time-consuming step in the process of applying an ecosystem service approach. The aim of this step should be to produce predictive models for ecosystem service provision, which describe the relationships between hydrology (or hydrological habitat types) and the value of each service provided. These models should take into account not only the environmental conditions on the floodplain site, but also the surrounding landscape and the functional connectivity of the floodplain to other wetland resources. Ideally, uncertainty in the predictions should also be quantified, for example through confidence intervals derived from regression or bootstrap methods.

5. A number of potential management strategies (including a continuation of the current management regime) should be envisaged, and the likely ecosystem services provided by each scenario should be predicted using the ecosystem service models. Decision makers may either choose to explore the range of possibilities using optimisation methods (Chapter 6), or may instead focus on a smaller number of proposed alternatives (Chapter 7).

6. The information describing the impacts of each management scenario on the provision of multiple ecosystem services must be translated successfully to decision makers. In particular, the trade-offs between services, and the degree to which it is possible to provide multifunctionality, should be described. If the indicators are well chosen and the management scenarios are well-defined then this should not be too difficult, but care should be taken to avoid complex or discipline-specific language and figures. Data are commonly presented in ecosystem service research as radar plots (MEA 2005, Raudsepp-Hearne et al. 2010a), line plots (Sanon et al. 2012), or tables (Chapter 7). Qualitative explanations of each ecosystem service, as used in the Chapter 7 survey, and maps of the study floodplain (Troy and Wilson 2006) may help to improve the general understanding of decision makers. If the data allow, descriptions of the uncertainty surrounding each prediction should also be provided.

7. Finally, the responses of decision makers to the supplied ecosystem service information should be recorded and analysed to ensure that the ecosystem service information has not been misunderstood. If necessary, the previous six steps should be repeated to supply additional information. The information generated during the project should be shared online for use in future meta-analyses or benefit transfer

studies, for example through the Environmental Valuation Reference Inventory (EVRI 2014).

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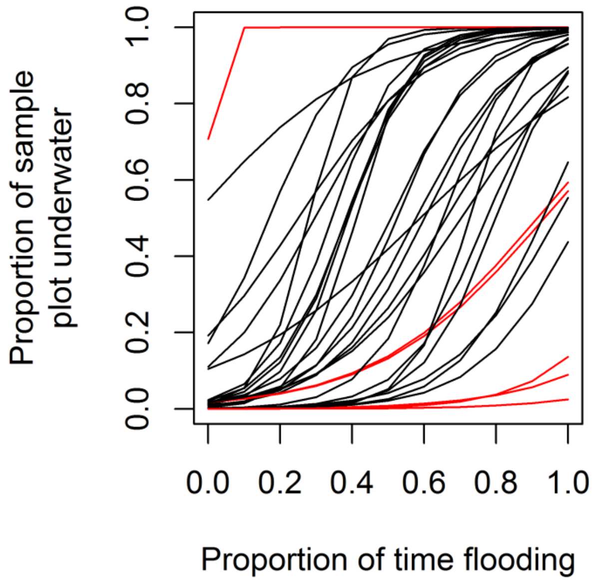
Appendices

A. Appendix to Chapter 2

A.1. Topographic height of the four flood entry points shown in Figure 2.7 over the five years of the study. Heights were estimated from the topographic dataset described in Chapters 2 and 4. The topographic measurements of the site in 2013 are visualised in Figure 2.7.

Flood entry point	Height in 2009 (m)	Height in 2010 (m)	Height in 2011 (m)	Height in 2012 (m)	Height in 2013 (m)
a	2.6	2	2	2	2
b	5.3	3.7	3.7	3.7	3.7
c	5	5	5	2.5	2.5
d	5	3.5	3.5	3.5	3.5

B. Appendix to Chapter 3



B.1. Logistic regression models that predict the hydrology of each sample plot from river level data. The proportion of the sample plot underwater was measured on a given occasion, and the proportion of time during the preceding two weeks that flood water was entering the site was used as the explanatory variable. Each line shows the relationship for one sample plot (n=28). Red lines indicate non-significant (at alpha = 0.05) relationships, black lines indicate significant ones.

B.2. List of species that were recorded in the sample plots for the first time in 2011 and 2013.

Species	Year of colonisation	Number of sample plots colonised	Broad habitat
Leontodon autumnalis	2011	1	Terrestrial
Hordeum secalinium	2011	1	Terrestrial
Callitriche stagnalis	2011	2	Aquatic
Elodea nuttallii	2011	1	Aquatic
Hippuris vulgaris	2011	1	Aquatic
Potamogeton natans	2011	1	Aquatic
Potamogeton pusillus	2011	1	Aquatic
Filamentous algae spp.	2011	1	Aquatic
Plantago lanceolata	2011	2	Terrestrial
Sisymbrium officinale	2011	1	Terrestrial
Epilobium montanum	2011	1	Terrestrial
Veronica serpyllifolium	2011	1	Terrestrial
Iris pseudocorus	2011	1	Aquatic
Veronica scutellata	2013	3	Terrestrial
Aethusa cynapium	2013	2	Terrestrial
Eleocharis palustris	2013	2	Terrestrial/ emergent
Hypocharis radicata	2013	2	Terrestrial
Viola palustris	2013	2	Terrestrial
Chenopodium album	2013	1	Terrestrial
Cerastium arvensis	2013	2	Terrestrial
Sinapsis arvensis	2013	2	Terrestrial

B.3. List of species which went extinct from the survey quadrats, either between 2009 and 2011, or between 2011 and 2013.

Species	Year of no record	Number of sample plots extinct	Broad habitat
Epilobium hirsutum	2011	1	Terrestrial
Galium aparine	2011	4	Terrestrial
Lathyrus pratensis	2011	1	Terrestrial
Rumex acetosa	2011	2	Terrestrial/ emergent
Phalaris arundinacea	2011	1	Aquatic
Quercus robur	2011	1	Terrestrial
Veronica catenata	2011	1	Terrestrial/ emergent
Arctium minus	2011	1	Terrestrial
Anthriscus sylvestris	2013	2	Terrestrial
Carex otrubae	2013	1	Terrestrial
Cirsium vulgare	2013	8	Terrestrial
Juncus articulatus	2013	1	Terrestrial
Matricaria discoidea	2013	1	Terrestrial
Myosotis scorpioides	2013	1	Terrestrial/ emergent
Leontodon autumnalis	2013	1	Terrestrial
Hippuris vulgaris	2013	1	Terrestrial/ emergent
Potamogeton pusillus	2013	1	Aquatic
Sisymbrium officinale	2013	1	Terrestrial
Epilobium montanum	2013	1	Terrestrial
Veronica serpyllifolia	2013	1	Terrestrial
Iris pseudacorus	2013	2	Terrestrial/ emergent

C. Appendix to Chapter 5

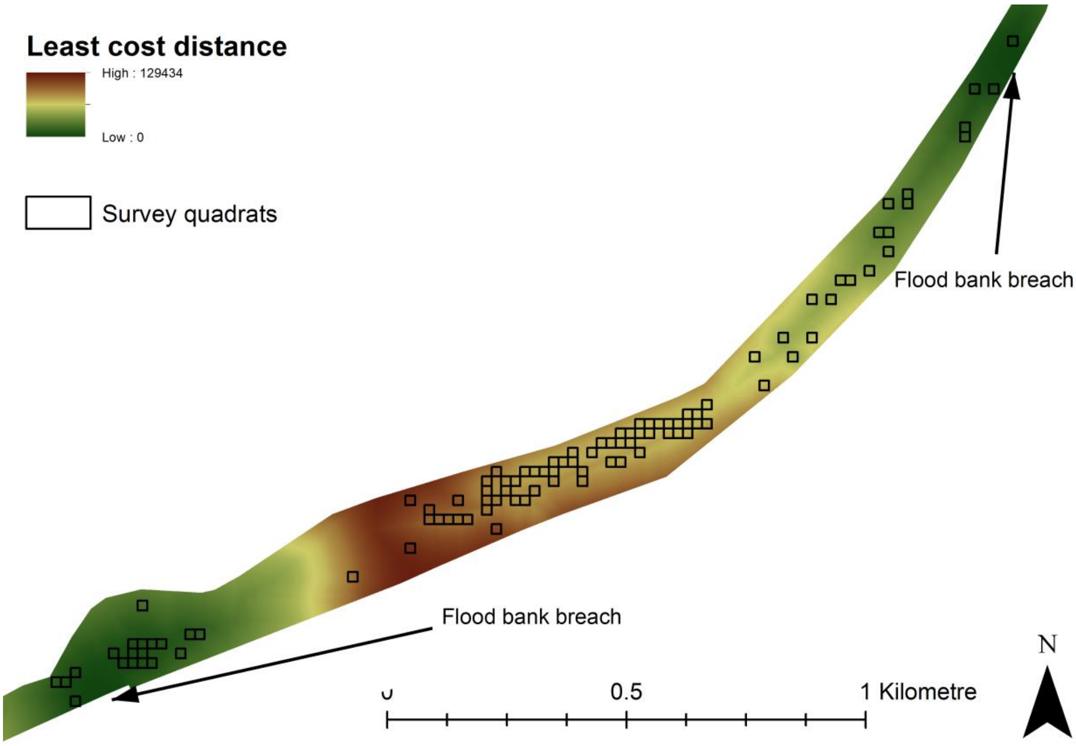
C.1. Counts of debris items in the River Don at Fishlake

This supplementary information describes the data collection and analysis for Figure 5.1. Figure 5.1 is an additional figure produced for Chapter 5 of this thesis, and is not part of the accompanying manuscript.

Debris items were counted from the riverbank on 68 occasions between the 6/6/2011 and the 19/4/2013. On each occasion, debris items that passed the location of the surveyor were counted for a period of five minutes and the direction of river flow (either upstream or downstream) was recorded. Sticks (small tree branches), logs (tree branches that were approximately 10 cm or more in diameter), man-made items, and agglomerations of floating rush were recorded separately, but were pooled for analysis. The size of larger agglomerations was estimated visually, and the number of debris items was estimated as the number of 30 cm diameter discs that would be needed to cover the agglomeration. The difference in total quantity of debris between occasions of upstream and downstream flow was analysed using a Welch t-test.

Over the course of all surveys, 985 debris items were recorded. Sticks made up the majority of the debris (59%), followed by rush (18%), logs (13%) and man-made items (10%). The mean quantity of debris was 8 items when the direction of flow was downstream, and was 55 items when the direction of flow was upstream. The quantity of debris in the river during periods of upstream flow was significantly greater than the quantity during downstream flow ($t = 3.16$, $df = 8.12$, $p = 0.01$).

D. Appendix to Chapter 6



D.1. Quadrat locations for the surveys used in Chapters 5 and 6, and the least cost variable that was used as an index of the shortest wetted distance from the two bank breaches. The least cost measure was used as an index of the probability that items brought in by flood events would travel to different parts of the floodplain. The cost distance was calculated from the two flood bank breach locations, and the cost surface was specified as the number of occasions that each part of the floodplains was dry during the course of the water level monitoring described in Chapters 2 and 4. The minimum value of this cost surface was 0, indicating that the location was always submerged, and items could thus flow freely. The maximum value of the cost surface was 52, which would indicate that the location was never submerged and it would thus be unlikely that items would be carried over the area by flooding. Green indicates areas that are expected to be highly connected to the river, while brown indicates less connected areas.

D.2. List of floodplain grazing marsh and Doncaster region biodiversity action plan plant species.

Species	FGM indicator species	BAP – listed species
Achillea millefolium	x	x
Cirsium vulgare	x	x
Juncus articulatus	x	x
Myosotis.scorpiodes	x	x
Agrostis spp.	x	x
Alopecurus geniculatus	x	x
Cynosurus cristatus	x	x
Festuca rubra agg.	x	x
Glyceria fluitans	x	x
Lolium perenne	x	x
Phalaris arundinacea	x	x
Anthriscus sylvestris	x	
Bellis perennis	x	
Cerastium fontanum	x	
Cirsium arvense	x	
Equisetum arvense	x	
Epilobium hirsutum	x	
Galium aparine	x	
Juncus effusus	x	
Juncus inflexus	x	
Lathyrus pratensis	x	
Persicaria.amphibia	x	
Lotus corniculatus	x	
Plantago major	x	
Potentilla anserina	x	
Potentilla reptans	x	
Ranunculus acris	x	
Ranunculus repens	x	
Rumex conglomeratus	x	
Rumex obtusifolius	x	
Rumex acetosa	x	
Senecio jacobaea	x	
Taraxacum officinale	x	
Trifolium pratense	x	
Trifolium repens	x	
Urtica dioica	x	
Arrhenatherum elatius	x	
Alopecurus pratensis	x	
Bromus hordeaceus	x	
Elymus repens	x	
Dactylis glomerata	x	

D.2. (continued)

Species	FGM indicator	BAP listed
Deschampsia cespitosa	x	
Festuca arundinacea	x	
Holcus lanatus	x	
Rumex crispus	x	
Geranium dissectum	x	
Trifolium dubium	x	
Leontodon autumnalis	x	
Hordeum secalinum	x	
Plantago lanceolata	x	
Iris pseudacorus	x	
Eleocharis.palustris	x	
Heracleum sphondylium	x	
Lemna minor	x	
Hypochoeris radicata	x	
Leucanthemum vulgare	x	
Veronica scutellata	x	
Capsella bursa-pastoris	x	
Rorippa nasturtium-aquaticum	0	x
Glyceria maxima	0	x
Phleum bertolonii	0	x
Veronica catenata	0	x
Potamogeton natans	0	x
Lemna trisulca	0	x
Poa annua	x	

D.3. Generalised linear models for each of the seven ecosystem services indicators.

D.3.1. Proportion of plant species that were floodplain grazing marsh indicators species

Coefficient	Estimate	Standard error	z	p
Intercept	2.10	0.5	4.17	<0.001
Flood exposure	-0.05	0.01	-3.27	0.001

D.3.2. BAP species richness

Coefficient	Estimate	Standard error	z	p
Intercept	1.48	0.09	15.82	<0.001
Flood exposure	-0.15	<0.01	-4.42	<0.001

D.3.3. Probability of cow dung presence

Coefficient	Estimate	Standard error	z	p
Intercept	1.27	0.3	4.28	<0.001
Flood exposure	-0.08	0.01	-6.78	<0.001

D.3.4. Probability of odonata presence

Coefficient	Estimate	Standard error	z	p
Intercept	-2.65	0.44	-5.97	<0.001
Flood exposure	0.14	0.27	5.38	<0.001
Flood exposure squared	-0.001	<0.01	-4.02	<0.001
Vegetation height	0.03	0.02	1.81	0.07
Spatial predictor	9.12	1.25	7.28	<0.001

D.3.5. Probability of debris presence

Coefficient	Estimate	Standard error	z	p
Intercept	2.23	<0.01	3.12	0.002
Flood exposure	<0.01	<0.01	-2.88	0.004
Vegetation height	<0.01	<0.01	1.88	0.06
Flow distance from nearest flood entry point	-1.34	<0.01	-1.91	0.06

D.3.6. Volume available for water storage

Coefficient	Estimate	Standard error	z	p
Intercept	4244.37	175.9	24.12	<0.001
Flood exposure	153.67	16.89	9.09	<0.001
Flood exposure squared	-1.51	0.31	-4.84	<0.001
Spatial predictor	-4003.66	702.64	-5.7	<0.001

D.3.7. Probability of bird presence

Coefficient	Estimate	Standard error	z	p
Intercept	-4.6	0.65	-7.06	<0.001
Flood exposure	0.17	0.03	4.57	<0.001
Flood exposure squared	<0.01	<0.01	-3.97	<0.001
Vegetation height	-0.04	0.01	-2.38	0.018

E. Appendix to Chapter 7

E.1. Example question set for the simple treatment group.

Option A	Option B
Downstream village will flood every 22 years	Downstream village will flood every 25 years
Water voles live here	No water voles present

E.2. Example question set for the intermediate treatment group.

Option A	Option B
Downstream village will flood every 22 years	Downstream village will flood every 25 years
Water voles live here	No water voles present
It is not known how suitable this land is for cattle	It is not known how suitable this land is for cattle
Unknown number of wetland bird species present	Unknown number of wetland bird species present

E.3. Example question set for the complex treatment group.

Option A	Option B
Downstream village will flood every 22 years	Downstream village will flood every 25 years
Water voles live here	No water voles present
Herd of 0 cows present	Herd of 20 cows present
4 wetland bird species live here	1 wetland bird species live here

E.4. Bias assessment statements.

F1	Flooding could directly affect me in the near future
F2	Flooding could directly affect people close to me in the near future
V1	I was aware of water vole conservation issues before today
V2	Water vole conservation is a worthy cause
B1	I like to see wetland birds
B2	I sometimes go out into nature specifically to experience wildlife
C1	It is important that beef production in the UK does not decrease in the future
C2	I am an active member of the farming community

F. Description of data appendices contained on the attached Compact Disc

F.1. Plant community dataset analysed in Chapter 3. This dataset includes records of 28 sample plots that were repeatedly surveyed in 2009, 2011, 2012, and 2013. Date, site number, transect, and records of 96 taxa are included. DAFOR-scale abundances are given numerically; Not present = 0, Rare = 1, Occasional = 2, Frequent = 3, Abundant = 4, Dominant = 5. In comma separated volume (csv) format.

F.2. Water vole presence/ absence dataset and environmental explanatory variables used in Chapter 4. This dataset contains data for 34 survey raft locations. The columns refer to (in order); the site number, range in water depth, water body width, range in water depth, largest flood magnitude, plant community diversity, vegetation height, number of visits to the raft when water voles were absent in 2011, number of visits to the raft when water voles were absent in 2012, number of visits to the raft when water voles were present in 2011, and number of visits to the raft when water voles were present in 2012. In comma separated volume (csv) format.

F.3. Odonate and debris quantities and environmental explanatory variables used in modelling for Chapter 5. This dataset contains data for 100 survey quadrats. The columns refer to (in order); the quadrat number, odonate abundance on the 18/07/12, odonate abundance on the 23/07/12, odonate abundance on the 03/08/12, odonate abundance on the 18/07/13, odonate abundance on the 16/07/13, odonate abundance on the 24/07/13, total debris quantity in June 2012, debris quantity observed from the footpath in June 2012, total debris quantity in August 2012, debris quantity observed from the footpath in August 2012, total debris quantity in June 2012, debris quantity observed from the footpath in June 2012, total debris quantity in May 2013, debris quantity observed from the footpath in May 2013, flood exposure, vegetation height, and least cost distance to the nearest flood entry point. In Excel 201 (xlsx) format.

F.4. Plant community dataset analysed in Chapter 6. This dataset includes records of surveys at 100 quadrats plots that were surveyed in 2012. Quadrat number and records of 96 taxa are included. DAFOR-scale abundances are given; Not present = blank, Rare = R, Occasional = O, Frequent = F, Abundant = A, Dominant = D. In Excel 2010 (xlsx) format.

F.5. Ecosystem service indicator an environmental explanatory variable dataset used in Chapter 6. This dataset includes records of surveys at 100 quadrats plots that were surveyed in 2012. The columns refer to (in order); the quadrat number, plant community species richness, number of FGM indicator species, number of BAP priority spceies, number of times odonates were encountered in the quadrat during surveys, number of times debris items were encountered, number of times cowpats were encountered, mean available flood storage volume in the quadrat over 52 survey occasions, flood exposure, mean vegetation height at five points in the quadrat (cm) least cost distance from the nearest flood entry point, northing of the quadrat centroid (OS British National Grid), easting of the quadrat. In Excel 201 (xlsx) format.